Calcasieu Estuary Remedial Investigation/Feasibility Study (RI/FS): Baseline Ecological Risk Assessment (BERA)

Appendix H1: Assessment of Risks to Sediment-Probing Birds in the Calcasieu Estuary

Prepared For:

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Under Contract To:

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Prepared – October 2002 – By:

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Under Contract To:

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Appendix H1. Assessment of Risks to Sediment-Probing Birds in the Calcasieu Estuary

1.0 Introduction

Development and industrialization in and around the Calcasieu estuary in southwestern Louisiana in recent decades has led to concerns of environmental contamination in the area. A Remedial Investigation/Feasibility Study (RI/FS) was commissioned to determine the risks posed by environmental contamination to ecological receptors inhabiting key areas of the Calcasieu Estuary. A Baseline Ecological Risk Assessment (BERA) is required to meet this objective. This Appendix is part of the BERA and is conducted in accordance with the procedures laid out by the USEPA in the *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessment* (USEPA 1997a). Under the eight-step process described by the USEPA for conducting a BERA, a screening ecological risk assessment (SERA) must first be conducted to determine preliminary estimates of exposure and risk.

The SERA for the Calcasieu Estuary (CDM 1999) identified areas of concern, contaminants of concern (COCs), and ecological receptors potentially at risk. The SERA findings were revisited in a Baseline Problem Formulation (BPF) to yield a refined list of COCs, areas of interest, and ecological receptors to be considered in the BERA. The Phase II data collection provided more information and, therefore, a better tool to estimate risk at a screening level. Using this information, a conservative, deterministic assessment was conducted and can be found in Appendix G along with a description of the methods used to identify the COCs and areas of concern for sediment-probing birds.

This Appendix is organized as follows. Section 1 provides a brief overview of the results of the conservative, deterministic ERA for sediment-probing birds in detail in Appendix G. The Areas of Concern (AOCs) and COCs that screened through the conservative, deterministic assessment for sediment-probing birds are described in this section. Section 1 also includes a description of the conceptual model for sediment-probing birds in the Calcasieu Estuary. A statement outlining the purpose of this assessment concludes Section 1.

Section 2 describes the probabilistic risk assessment methods used to estimate risks of COCs to sediment-probing birds in the Calcasieu AOCs. Section 3 describes the probabilistic risk assessment results and Section 4 identifies the sources of uncertainty that could influence the estimated risks for sediment-probing birds. The final section of this Appendix, Section 5, contains the conclusions regarding risks of COCs to sediment-probing birds in the Calcasieu Estuary.

1.1 Deterministic Ecological Risk Assessment Summary

The methods and results of the deterministic ecological risk assessment are presented in detail in Appendix G. In summary, the deterministic assessment used a conservative approach to estimate risk to sediment-probing birds from chemicals of potential concern (COPCs) in the Bayou d'Inde, Upper Calcasieu River, and Middle Calcasieu River Areas of Concern (BI AOC, UCR AOC, MCR AOC, respectively). Several reference sites, including Bayou Connine Bois and Choupique Bayou, were also included in the deterministic assessment to provide a basis of comparison of risks. Those were called reference areas in this assessment. The deterministic assessment compared potentially attainable high exposures with conservative adverse effects benchmarks to provide a means of identifying which contaminants are a potential concern to sediment-probing birds and in which areas of the Calcasieu Estuary system. Generally, a risk quotient (total daily intake/effect dose) for sediment-probing birds greater than one for any COPC in any of the Calcasieu areas resulted in the COPC being passed through to the probabilistic ecological risk assessment. COPCs that screened through the SERA are now referred to as COCs. Selenium, lead, and mercury were screened through for all AOCs whereas TCDD and equivalents (TCDD-TEQs) were screened through for BI AOC and UCR AOC. The reference areas were also screened through to the probabilistic risk assessment so that risks. Results of the deterministic risk assessment are presented in Table H1-1.

1.2 Contaminants of Concern

The COCs that screened through included mercury, TCDD-TEQs, lead, and selenium and are described below.

Mercury

Mercury is found in the environment as the metal, Hg^0, and as divalent mercuric Hg(II) species. In the water column, Hg^0 is oxidized to Hg(II) under acidic conditions. Hg(II) undergoes a number of important reactions, one of which is methylation by microbes and adsorption and absorption by biota (Stein *et al.* 1996). Biomethylation occurs both in the sediments, where sulfate-reducing bacteria are the primary methylators of mercury, and in the water column (Winfrey and Rudd 1990). Methylation in the water column also occurs abiotically, mediated by dissolved organic carbon (Weber 1993). Methylmercury may make up as much as 25 percent of the mercury in rivers and lakes (Gilmour and Henry 1991).

Methylmercury is highly soluble in water, extremely mobile, and thus readily enters the aquatic food web. Because methylation is higher under anaerobic conditions, benthic organisms in the anaerobic zones of sediment may be exposed to high methylmercury concentrations. These organisms are consumed by a variety of species, including sediment-probing birds, leading to biomagnification up the food chain. The accumulation of methylmercury in aquatic organisms has been well documented, with concentrations in carnivorous fish 10,000 to >1,000,000 times the concentrations found in ambient waters (Stein *et al.* 1996). Gilmour and Henry (1991) showed that fish from contaminated systems may continue to contain high levels of methylmercury long after inputs to the systems have ceased. Also, the efficient assimilation of the lipophilic methylmercury in fat and muscle and the lack of elimination results in increasing methylmercury concentrations with the age and size of fish and wildlife predators.

This assessment focuses on the risks posed by methylmercury to sediment-probing birds because this species of mercury is more readily bioaccumulated and more toxic to wildlife than is metallic mercury. Further, previous assessments of methylmercury risks to wildlife have shown that species higher in the aquatic food chain are at particular risk of experiencing adverse effects, including reduced reproduction, impaired growth and development, and death (MacIntosh *et al.* 1994; USEPA 1997b; Moore *et al.* 1999). Sediment-probing birds are fairly high in the food chain and are potentially at high risk of exposure to methylmercury because they consume sediment-dwelling invertebrates as well as sediments via incidental ingestion.

Lead

Lead and its salts are generally poorly soluble in water and tend to partition into sediment as organic complexes (WHO 1989). As such, lead bioavailability in natural systems is limited and almost all of the lead is tightly bound to the sediment. In aquatic systems, lead is uptaken by primary producers and consumers at a rate strongly controlled by bioavailability of lead. Uptake by aquatic organisms is also controlled by temperature, salinity, pH, humic acid levels, and alginic acid concentrations (WHO 1989).

Consumers may take up large quantities of lead via their food, but the ingested metal might not be fully bioavailable and, therefore, actual exposure will be low. Bioavailability is reduced in presence of organic matter, sediment, and mineral particles (WHO 1989). Bioaccumulation is usually not a problem with this element. In fish, accumulation equilibrium is reached after only several weeks of exposure and lead tends to deposit in gills, liver, kidney, and bones. Fish eggs are also associated with some lead deposition, but mostly on egg surfaces (WHO 1989).

In general, inorganic lead compounds exhibit much lower toxicity than the trialkyl- and tetraalkyllead compounds. The latter compounds are more water soluble and are rapidly taken up and eliminated by aquatic organisms. Lead shot taken up by birds into their gizzards is an important source of lead poisoning (WHO 1989). Lead shot is quickly ground down (few weeks) in the gizzard to lead powder, which is very toxic. Some birds are very sensitive and one ingested pellet is sufficient to cause mortality (WHO 1989).

Lead's toxic mode of action as a metabolic poison affects a broad range of tissues and organs (Kendall *et al*. 1996). The alternation of enzyme function causes severe degeneration in the central nervous system, blood production (inhibits iron activity), and kidney activity (Kendall *et al*. 1996). Lead also affects reproduction including premature abortions and mortality of neonates. Chronic exposure to lead is associated with loss of body weight, behavioral changes, and partial paralysis (Sanderson and Belrose 1986; Heitmeyer *et al*. 1993).

This assessment focuses on the risks posed by lead to sediment-probing birds because lead can be toxic when ingested in large quantities. Lead is not expected to bioaccumulate in prey items consumed by sediment-probing birds. Thus, exposure via this route is not likely to be extensive. Exposure of birds to heavily contaminated sediments via incidental ingestion along with food can be very high.

Selenium

The fate of selenium and its compounds in the environment is influenced to a large degree by its oxidation state. The valence states of selenium range from -2 (hydrogen selenide) through 0 (elemental selenium), +2 (selenium dioxide), +4 (selenite) and +6 (selenate). The behavior of various compounds of selenium in the environment depends on ambient conditions including pH, the presence of metal oxides and biological activity (ATSDR 1996; Maier *et al.* 1988).

Elemental selenium is essentially insoluble and will remain inert when released in the environment under anaerobic conditions. Heavy metal selenides and selenium sulfides predominate in acidic soils and soils with high organic matter, and will remain insoluble and immobile in this form (NAS 1976). Selenites and selenates are water soluble and are, therefore, more bioavailable in surface water and water contained in soils (Eisler 2001; ATSDR 1996; Robberecht and Van Grieken 1982). In general, these mobile forms of selenium dominate under aerobic and alkaline conditions. Sodium selenate is one of the most mobile selenium compounds in the environment because of its high water solubility and inability to adsorb onto particulates (NAS 1976). Selenium bioconcentrates and biomagnifies in aquatic food chains from invertebrates to birds (Ohlendorf *et al*. 1986a; 1986b; Lemly 1985; Saiki and Lowe 1987). Lemly (1985) reported BCFs of 1,500-1,850 and BAFs of 1,746-3,975 for selenium in freshwater species. Concentrations of selenium in river otter and raccoon have been measured (wet weight) in various organs ranging from 0.2 to

2.8 mg Se/kg (Wren 1984). These studies demonstrate that selenium has the potential to biomagnify up the food chain and accumulate in sediment-probing birds.

This assessment focuses on the risks posed by selenium to sediment-probing birds because this contaminant is expected to biomagnify up the food chain. Selenium bioconcentrates in aquatic food chains from invertebrates to birds with diet being identified as the primary source for fish and sediment-probing birds having the highest body burdens among avians (Eisler 2001). Sediment-probing birds are fairly high in the food chain and are potentially at high risk of exposure to selenium because they consume invertebrates as well as sediments directly via incidental ingestion while foraging.

TCDD-TEQs

Tetrachlorinated dibenzo-*p*-dioxin and equivalents represent a group of aromatic compounds with similar properties (WHO 1989). The term equivalents refers to a specific group of polychlorinated dibenzo-*p*-dioxin (PCDDs) congeners, polychlorinated dibenzofuran (PCDFs) congeners and polychlorinated biphenyl (PCB) congeners. This group has a common structural relationship that includes lateral halogenation and the ability to assume a planar conformation. The planar conformation is important as it leads to a common mechanism of action in many animal species that involves binding to the aryl hydrocarbon (Ah) receptor and elicitation of an Ah receptor-mediated biochemical and toxic response (van den Berg *et al.* 1998; Safe 1994).

Each of these compounds, while similar in structure and acting at the same receptor, has different potencies, depending on the individual congener. To address these issues and effectively estimate the relative toxicity of these mixtures, a system has been created involving the development and use of toxic equivalency factors (TEFs).

This approach is based on the *in vivo* and *in vitro* toxicity of each of the compounds in relation to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD). TCDD is considered to be most toxic member of the this class of chemicals (van den Berg *et al.* 1998; Birnbaum and DeVito 1995; Safe 1994) and the toxicity of the others depends on the degree of chlorination, the chlorination sites, and the ability to achieve a planar form, relative to TCDD. There are a number of assumptions made when using the TEF approach. These include: 1. the congeners are Ah-receptor antagonists and their toxicological potency is mediated by their binding affinity, and 2. no interaction occurs between the congeners and thus the sum of the individual congener effects accounts for the potency of the mixture. The overall effect of these assumptions is a potency estimate or toxic equivalent (TEQ) value. A more detailed discussion of the TEF approach for expressing the toxicity of this class of chemicals is presented in Appendix G.

The environmental degradation and metabolism of the congeners varies due to their unique physical/chemical properties. These can cause substantial differences between the congeners detected in environmental samples and the congener makeup of the original product (van den Berg *et al.* 1998). The majority of these congeners have low solubility, low vapor pressure and high resistance to chemical breakdown, and are, therefore, highly persistent in the environment. They are also highly lipophilic with a high propensity to bind to organic and particulate matter. When released to aquatic systems, the majority of these compounds form associations with dissolved and/or particulate matter in the water column; biodegradation is considered to be a relatively minor fate process in water (NRCC 1981; Howard *et al*. 1991). Aquatic sediments provide a sink for these compounds and may represent long term sources to the aquatic food web (Kuehl *et al.* 1987; Corbet *et al.* 1983; Tsushimoto *et al.*; 1982; Muir *et al.* 1985). As sediments are resuspended and carried downstream, they tend to accumulate in areas where currents are slow and the particles have time to settle.

Organisms may be exposed to TCDD and equivalents through trophic transfer. PCDDs, PCDFs and PCB congeners are highly bioaccumulative contaminants that increase in concentration as they are passed up the food chain (i.e., biomagnification). For organisms inhabiting the Lake St. Clair ecosystem, Haffner *et al.* (1994) observed that PCB concentrations increased from 935 ng/kg in sediments, to 1.36 mg/kg in bivalves, to 7.24 mg/kg in oligochaetes, and to 64.9 mg/kg in predatory gar pike. Mink are particularly sensitive to PCBs and similar chemicals (Moore *et al.* 1999). Research has found that they accumulate PCBs in their subcutaneous fat at levels 38 to 200 times dietary concentrations, depending on the PCB congener (USEPA 1993). The avian predators of the Calcasieu estuary study area would similarly be expected to accumulate PCBs from the prey they consume.

This assessment estimates the risks posed by coplanar congeners to sediment-probing birds because these compounds are expected to biomagnify up the food chain. Further, previous assessments have shown that species higher in the aquatic food chain are at particular risk of experiencing adverse effects, including reduced reproduction, impaired growth and development, and death (Moore *et al.* 1999; Tillitt *et al.* 1996; Heaton *et al.* 1995). Sediment-probing birds are moderately high in the food chain and are potentially at high risk of exposure to coplanar congeners because they consume invertebrates and sediments.

1.3 Receptors of Concern

Thorough observations of the study area led to the identification of many sediment-probing bird species including willet, spotted sandpiper, and black-necked stilts (Chemrisk 1996). The named species are commonly observed in the study area and are opportunistic feeders that may consume aquatic invertebrates and fish as parts of their diets. The following sections review the life histories and foraging behaviors of these three species. This information is then used to develop the life history and foraging behavior of a hypothetical receptor. The exposure assessment for sediment-probing birds exposed to COCs will be based on the hypothetical receptor that incorporates many of the characteristics typical of the listed species rather than focusing on any particular species.

Willet (*Catoptrophorus semipalmatus*)

The willet is a pigeon-sized bird. The sexes are very similar in size, females being slightly heavier than males. Knopf (1977) reported weights between 200 and 300 g. Dunning (1993) reported willet weights of approximately 215 g, and the Connecticut Department of Environmental Protection gives a range of 198 to 340 g (CDEP 1999). These birds are about 38 cm in length (Dunning 1993). Willets breed locally in Canada, the United States, and the West Indies, wintering in the southern United States and South America (Knopf 1977; USGS 2001). Willets are year-round inhabitants of the Calcasieu Estuary (McLaren/Hart-Chemrisk 1998).

Willet habitat includes coastal marshes, sand dunes, mud flats and rocky areas (Hayman *et al.* 1986). Willets nest and feed in separate locations. When nesting, willets often form small, loose, breeding colonies. Willets feed on sandbars, mud flats, and along tidal creeks and pannes of salt marshes.

Willets primarily feed on aquatic invertebrates, as well as lesser quantities of fish. Prey include crustaceans, molluscs, marine worms, aquatic insects, and small fish (Ehrlich *et al.* 1988; Gough *et al.* 1998). The willet catches prey by pecking from surface water and probing sediment with it's bill (Hayman *et al.* 1986).

Black-necked Stilt (*Himantopus mexicanus*)

Black-necked stilts are medium sized shorebirds that measure 33 to 40 cm in length with a mean body weight of 166 g (Hamilton 1975). Black-necked stilts are found in the southern and western United States, year round in the Calcasieu Estuary, and as far south as Peru (Knopf 1977; Hayman *et al.* 1986; McLaren/Hart-Chemrisk 1998).

Habitat preferences of black-necked stilts include coastal salt marshes, commercial salt pannes, inland saltwater and freshwater lakes, mudflats, grassy marshes, and sewage farms. Nesting takes place in small colonies.

Black-necked stilts are visual feeders, foraging primarily for aquatic invertebrates in the top 20 cm of the water column. They also eat fish, reptiles, and amphibians. Prey include, but are not limited to, brine flies, brine shrimp, crayfish, snails, tadpoles, and seeds (Ehrlich *et al.* 1988; Gough *et al.* 1998).

Spotted Sandpiper (*Actitis macularia*)

The spotted sandpiper is a small shorebird commonly found in Northern Alaska, Canada, and the southern United States (USEPA 1993). Coastal areas in the southern United States, including Louisiana, serve as wintering grounds, while northern regions are used for breeding (USEPA 1993; McLaren/Hart-Chemrisk 1998). Oring and Lank (1986) found a significant difference in weight between sexes (males 40 g, females 50 g). Maxson and Oring (1980) similarly reported a mean adult female weight of

47 g (range 43 to 50 g) and adult male weight of 38 g (range 34 to 41 g). An average weight of 40 g for both sexes was reported in Dunning (1984; standard deviation 6.15, range 30 to 60 g).

Spotted sandpipers inhabit areas along the edges of bodies of water. Inland habitats include lakes, ponds, and rivers. They also inhabit coastal environments, where they search beaches, inlets, and creeks for food (USEPA 1993). Sandpipers prefer to nest in semi-open vegetation. Nests are usually well concealed on the ground, lined with vegetation and hidden by grasses, or among rocks and driftwood. While breeding, sandpipers seek the densest vegetation (USEPA 1993). Sandpipers will walk slowly along the shores of sandy beaches, and the muddy edges of inlets and creeks, picking up food along the way (USEPA 1993). Inland, sandpipers feed along the shores of sandy ponds, streams, and mountain torrents. They will sometimes stray into meadows, fields or gardens in agricultural areas, where they find their food in low vegetation or off the ground (USEPA 1993).

Spotted sandpipers generally feed on prey that swim in the first 4 cm of the water column. They feed almost exclusively on small invertebrates and small fish. The sandpiper has the ability to capture flying insects, however, it prefers to catch its prey by probing and gleaning it from substrate (USEPA 1993). Young sandpipers begin feeding themselves almost immediately after hatching (USEPA 1993).

Hypothetical Sediment-Probing Receptor of Concern

The hypothetical sediment-probing bird for this assessment is based upon the behavior and characteristics described for sandpipers, willets, and stilts.

- The receptor body weight is assumed to equal the average of the three species with a coefficient of variation equal to 15%. This value is a typical coefficient of variation found for wild birds. A "what if" scenario will also be considered, where the receptor body weight will be set to the weight of the smallest bird (spotted sandpiper) to consider species with the greatest potential for exposure (highest metabolic rate and food ingestion rate when normalized to body weight);

- The hypothetical receptor is assumed to have a relatively small foraging range with high site fidelity and no territoriality. BI AOC, MCR AOC, and UCR AOC regions of the Calcasieu Estuary system were identified as areas of concern for sediment-probing birds in the deterministic risk assessment. It is assumed that receptors will forage exclusively within these areas;
- The diet is assumed to consist entirely of small invertebrates; and,
- The hypothetical receptor is assumed to be resident year-round in the Calcasieu area. The temporal scale for this assessment is long term because: (1) contaminant levels are unlikely to exhibit temporal variability because of their high persistence, and (2) chronic toxicity typically occurs at much lower levels than acute toxicity.

1.4 Conceptual Model

The conceptual model illustrates the relationships between sources and releases of COCs, their fate and transport, and the pathways through which COCs reach sediment-probing birds exert and potential adverse effects. The model enhances the level of understanding regarding the relationships between human activities and ecological receptors at the site under consideration. In so doing, the conceptual model provides a framework for predicting effects on ecological receptors and a template for generating risk questions and testable hypotheses (USEPA 1997a;

1998a). The conceptual site model developed for the Calcasieu Estuary is described in greater detail in Chapter 7 of the BPF. It summarizes information on the sources and releases of COCs, the fate and transport of these contaminants, the pathways by which ecological receptors are exposed to the COCs, and the potential effects of these contaminants on the ecological receptors that occur in the Calcasieu Estuary. In turn, this information is used to develop a series of risk hypotheses that provide predictions regarding how ecological receptors will be exposed to and respond to the COCs.

Sediment-probing birds are exposed to a number of COPCs in the Calcasieu Estuary system and the deterministic risk assessment (Appendix G) identified those COCs that pose potential risks to these animals. Specifically, sediment-probing birds are at greatest risk from mercury, selenium, lead, and TCDD-TEQs. These contaminants are available for uptake by sediment-probing birds, primarily through the food chain and incidental ingestion of contaminated sediment. The Phase II sampling program provided data identifying substantial residues of these contaminants in aquatic invertebrates and sediments. Other routes of exposure, including inhalation, water consumption, and transdermal transfer have been excluded from this assessment as their contribution to overall exposure is likely negligible.

1.5 Assessment Endpoints

The assessment endpoint in this assessment is the survival, growth, and reproduction of sediment-probing birds.

An assessment endpoint is an ‘explicit expression of the environmental value that is to be protected’ (USEPA 1997a). The selection of assessment endpoints is an essential element of the overall ERA process because it focuses assessment activities on the key environmental values (e.g., reproduction of sediment-probing birds) that could be adversely affected by exposure to environmental contaminants. Assessment endpoints must be selected based on the ecosystems, communities, and species that occur, have historically occurred, or could potentially occur at the site (USEPA 1997a).

A measurement endpoint is defined as 'a measurable ecological characteristic that is related to the valued characteristic selected as the assessment endpoint' and it is a measure of biological effects (e.g., mortality, reproduction, growth; USEPA 1997a). Measurement endpoints are frequently numerical expressions of observations (e.g., toxicity test results, community diversity measures) that may or may not be compared to similar observations at a control and/or reference site.

To support the identification of key assessment and measurement endpoints for the Calcasieu Estuary BERA, the United States Environmental Protection Agency (USEPA) convened a BERA workshop in Lake Charles, LA on September 6 and 7, 2000. The workshop participants included representatives of the USEPA, United States Geological Service (USGS), National Oceanic and Atmospheric Administration (NOAA), Louisiana Department of Environmental Quality (LDEQ), United States Fish and Wildlife Service (USFWS) and CDM Federal. The workshop was designed to enable participants to articulate the goals and objectives for the ecosystem (i.e., based on the input that had been provided by the community in a series of public meetings), to assess the state of the knowledge base, to define key issues and concerns, and to identify the chemicals and areas of potential concern in the study area. This workshop provided a basis for refining the candidate assessment endpoints that had been proposed based on the results of the SERA (CDM 1999). Workshop participants also identified a suite of measurement endpoints that would provide the information needed for evaluating the status of the assessment endpoints (MacDonald *et al.* 2000).

1.6 Measurement Endpoints

The potential for adverse effects on this foraging guild will be evaluated using prey tissue and sediment residue data. Specifically, the data on the concentrations of contaminants measured in aquatic invertebrates (<12.5cm in length) and in sediments from each subarea of concern will be used. These data will be compiled by geographic area within the estuary (based on the diet and foraging range of a hypothetical sediment-probing receptor), incorporated into a daily intake exposure model, and compared to appropriate toxicity values for survival and reproduction of avian species.

1.7 Risk Hypothesis and Questions

The following risk hypothesis was developed to identify the key stressor-effect relationships that will be evaluated in the probabilistic ecological risk assessment:

> Based on the physical-chemical properties (e.g., K_{ow}s) of the bioaccumulative contaminants of concern, the nature of the food web in the Calcasieu Estuary, and the effects that have been documented in laboratory studies, mercury, selenium, lead, and TCDD-TEQs released into surface waters accumulate in the tissues of aquatic organisms to levels that adversely affect the survival, growth, and/or reproduction of sediment probing birds.

To provide a basis for assessing ecological effects, the assessment endpoint must be linked to the measurement endpoint by risk questions. In this study, the investigation to assess the effects of environmental contaminants on the sediment-probing birds were designed to answer the following risk questions:

- Are the levels of contaminants in the tissues of prey species of sediment-probing birds in the Calcasieu Estuary sufficient to cause adverse effects on survival, growth, or reproduction?
- If yes, what are the probabilities of effects of differing magnitude for survival, growth, and/or reproduction of sediment-probing birds?

The linkages between the assessment endpoint and the measurement endpoints are articulated in greater detail in Table A1-21 of the BPF.

1.8 Purpose of Appendix

The purpose of this assessment is to test the above risk hypothesis by characterizing the risks posed to the sediment-probing birds associated with exposure to the COCs identified in Appendix G.

2.0 Methods

A step-wise approach was used to assess risks to the piscivorus bird community posed by the COCs in the Calcasieu Estuary. The steps in this process included:

- Collection, evaluation, and compilation of the relevant data on the concentrations of COCs in prey items in the Calcasieu Estuary;
- Assessment of exposure of sediment-probing birds to COCs (Figure H1-1);
- Assessment of the effects of COCs on sediment-probing birds (Figure H1-2); and,
- Characterization of risks to sediment-probing birds (Figure H1-3).

Each of these steps is described in the following sections of this report. The results of the deterministic assessment were briefly reviewed in Section 1.1. For details of this assessment, see Appendix G.

2.1 Collection, Evaluation, and Compilation of Data

Information on contaminant levels in prey tissues and sediments were collected in two phases, termed the Phase I and Phase II sampling programs. The Phase I program results indicated that the detection limits for many of the COPCs in tissues and sediments were orders of magnitude above corresponding benchmarks. Therefore, the Phase I results for tissues were not used in this assessment. The methods used to collect the tissue samples, quantify the levels of COCs, evaluate the reliability of the data, and compile the information in a form that would support the BERA are described in the following sections.

Sample Collection of Tissues **-** Data on levels of COCs in sediments and prey tissues (benthic invertebrates) were collected in two phases (Phase I and Phase II). Phase I data included more than 500 sediment samples and covered the time period between November, 1999 and March, 2000. The Phase II sampling program was designed to supplement the information collected in Phase I and to fill data gaps. The Phase II effort included more than 100 sediment samples. Sample collection utilized a stratified random sampling design and the number of samples taken at each sampling location was determined by the size of the area, previous contamination patterns, etc. Details of the sampling methods are included elsewhere (CDM 2000a; 2000b; 2000c; 2000d; 2000e).

The Phase I sampling program conducted in 2000 included some invertebrate tissue data. However, only large invertebrates were collected and analyzed for COCs. Large invertebrates such as adult crab could not be used in our assessment because adult crabs are not consumed by sediment-probing birds. Also, the detection limits for the organic chemicals were very high and there were many non-detects. This limited the utility of the dataset for estimating wildlife exposure.

For Phase II, more than 600 tissue samples were collected at sites located throughout the estuary between October, 2001 and November, 2001. Biota tissue samples were collected in three AOCs in the estuary (BI AOC, MCR AOC, and UCR AOC) and in the reference areas (Bayou Connine Bois, Calcasieu Lake, Choupique Bayou, Grand Bayou and Grand Bayou and Wetlands). There were also a number of sub-areas within the AOCs from which samples were taken. The USEPA Region V FIELDS tools were used to randomly select coordinates (i.e., latitude and longitude) for the assigned number of primary sampling stations and alternate sampling stations (i.e., which were sampled when it was not possible to obtain samples from the primary sampling stations). In the field, each sampling station was located with the aid of navigation charts and a Trimble differentially-corrected global positioning system (GPS). Using standard statistical power analysis methods, an evaluation of previously collected data was completed to determine the number of samples to be collected within each area and sub-area.

The methods used to collect, handle, and transport the tissue samples are described in CDM (2000a; 2000b; 2000c; 2000d; 2000e). Briefly, fish and invertebrate species were collected by hook and line, hand collection and netting. Each sample was wrapped in aluminum foil and put in a Ziploc® bag. All samples were kept frozen and shipped to laboratories in coolers on dry ice.

Chemical Analyses - Chemical analysis of sediment and tissue samples collected during Phase I and Phase II sampling programs was conducted at various contract laboratory program (CLP) and subcontract (non-CLP) analytical laboratories, including Quanterra-Severn Trent Laboratories, USEPA Region VI Laboratory, USEPA Region VI CLP laboratories, Olin Contract laboratories, PPG Industries contract laboratories, USGS Columbia Environmental Research Center, Texas A&M University, AATS, and ALTA Laboratories.

All tissue samples were analyzed for total target analyte list (TAL) metals, target compound list (TCL) semi-volatile organic compounds (SVOCs) and TCL pesticides. Total metals were quantified using the SW6010B method. Polycyclic aromatic hydrocarbons and/or other semi-volatile organic compounds were quantified using the SW8270C method. Methods SW8081A and SW8082 were used to quantify pesticides. Twenty percent of the tissue samples were analyzed for PCB congeners and dioxins/furans. EPA Method SW1668 was used to quantify PCB congeners and SW8290 was used for dioxins/furans.

EnChem laboratories used additional analytical methods to quantify mercury, polycyclic aromatic hydrocarbons (PAHs), pesticides and dioxins and furans. Methods 1631MOD and 1630MOD were used to quantify mercury and methylmercury, respectively. PAHs were quantified using Method 8270C-SIM. Method SW8082 and AXYS Method CL-T-1668A/Ver.3 were used to quantify pesticides. Dioxins and furans were quantified using AXYS Method DX-T-8290/Ver.2.

Data Validation and Verification - Obtained data were critically reviewed to determine their applicability to the assessment of risks to sediment-probing birds. This included validation of all of the sediment and tissue residue data contained in the database. Note that we were unable to confirm tissue data results against the original source.

Database Development - To facilitate data manipulation and analysis, a relational database was developed in MS Access format. All of the sediment chemistry and tissue residue data compiled in the database were georeferenced to promote mapping and spatial analysis using geographic information system (GIS)-based applications. The database was designed in such manner as to allow for a fully flexible data retrieval and analysis.

2.2 Probabilistic Ecological Risk Assessment

Monte Carlo analysis is an increasingly widely used approach to probabilistic risk assessment (USEPA 1997b; 1999). It is used to propagate uncertainty associated with the variability of input variables, as well as any incertitude associated with how to parameterize input distributions. In this assessment, we also use probability bounds analysis to determine the relative contributions of incertitude and variability to exposure estimates (see Chapter 9 of MacDonald *et al.* 2001 for more information on the uncertainty analysis approaches used here).

Monte Carlo analysis requires the specification of the statistical distributions of each of the input variables and their interdependencies as measured by correlations. Computer software such as Crystal Ball is used to 'sample' from these distributions and, via the exposure model equation, compute an exposure distribution. This process is repeated many times so as to build up a histogram that serves as the estimate of the full distribution of exposures (explicitly including the tail risks of extreme exposure).

Probability bounds analysis is an exact numerical approach (not based on simulation) that takes as input the same probability distributions used in Monte Carlo simulation, or, when they are difficult to specify precisely, bounds on these distributions (Ferson *et al.* 2002). The method then rigorously computes bounds on the cumulative distribution function. The spread between the bounds of an input or output distribution corresponds directly to the amount of incertitude we have about how to describe the variable. Probability bounds analysis is also useful when independence assumptions are untenable (such as between concentrations in sediments and benthic invertebrates), or when sparse empirical data make it difficult to quantify the correlations among variables.

2.2.1 Exposure Characterization

We estimate exposure of sediment-probing birds to methylmercury, selenium, lead, and TCDD-TEQs via a daily intake model. The exposure model calculates the total daily intake of COCs associated with the ingestion of food and sediments. Sediment-probing birds are unlikely to use the saline waters of the Estuary as a source of drinking water and the inhalation route of exposure has been shown to be an insignificant source of hydrophobic contaminants in previous assessments of the risks of these contaminants to aquatic-dependent wildlife (e.g., Moore *et al.* 1999). Chemical assimilation efficiency terms are not included in the exposure equation because the efficiencies of chemical adsorption in wild animals following ingestion will likely be similar to the efficiencies in laboratory animals in toxicity studies. Thus, the chemical assimilation efficiency terms will cancel out when the exposure and effect estimates are combined to estimate risk.

The temporal scale for this assessment is long term because: (1) levels of mercury, selenium, lead, and TCDD-TEQs are unlikely to exhibit high temporal variability, and (2) chronic toxicity occurs generally at lower levels than acute toxicity. The spatial scale of this assessment is considered to be consistent with home ranges reported for sediment-probing birds. The foraging area for the hypothetical receptor is set to 2,500 m^2. This area is equivalent to a circular zone of 56 metres in diameter or 250 metres of shoreline 10-m wide, both of which easily fit into each subarea of interest. This exposure assessment assumes that the hypothetical receptor is present year round in each of the identified Areas of Concern.

The exposure model is:

$$TDI = \frac{FMR \times \left(C_i + \left(C_s \times P_s\right)\right)}{AE_i \times GE_i} \qquad \textbf{EQUATION \#1}$$

where:

TDI = total daily intake of COCs (mg/kg BW/day),

C_i = concentration of COCs in invertebrates (mg/kg),

FMR = normalized free metabolic rate (Kcal/kg BW/day),

GE_i = gross energy of invertebrates (Kcal/kg prey),

AE_i = assimilation efficiency of invertebrates (unitless),

C_s = concentration of COCs in sediments (mg/kg), and

P_s = proportion of sediments ingested relative to diet (unitless)

Each input variable is described in detail below, including the parameterizations for the Monte Carlo analysis and the probability bounds analysis.

2.2.1.1 Selection of Criteria for Input Distributions

The distributions and distribution parameters used in the exposure analyses are summarized in Table H1-2. Input distributions were assigned as follows: lognormal distributions for variables that are positively skewed with a lower bound of zero and no upper bound (e.g., sediment and tissue concentrations), beta distributions for variables bounded by zero and one (e.g., prey assimilation efficiency), and normal distributions for variables that are symmetric and not bounded by one (e.g., body weight). The lognormal distribution is often used to provide good representations for physical quantities that are constrained to being non-negative, and that are positively skewed, such as contaminant concentrations, stream flows, or magnitudes of accidents (Small 1990). Ott (1995) provides an extensive discussion of the theoretical reasons for why contaminant concentrations in the environment are expected to be lognormally distributed. The beta distribution provides a flexible means of representing variability over a fixed range, such as zero to one (Small 1990). The beta distribution can take on a wide variety of shapes between the fixed endpoints and this flexibility has led to its empirical use in diverse applications. The normal distribution arises in many cases because of the central limit theorem, which results in a normal distribution for additive quantities such as body weights (Small 1990). The normal distribution can often be used even for variables that are non negative, as long as coefficients of variation (CV) are small (e.g., body weight). This is because many distributions converge to a normal distribution as CVs become small. With most random number generators, it is impossible to obtain numbers more than five standard deviations from the mean. Thus, as long as the CV is less than 0.2, there is no concern for selecting negative values for non-negative variables.

2.2.1.2 Input Distributions

Body Weight (BW)

Although body weight data are not used in the exposure model directly, they are a required variable in allometric models used to estimate free metabolic rate. For this assessment, we used body weights that represent an average-sized and a small-sized sediment probing bird (hypothetical receptor).

For the Monte Carlo analysis, the average body weight of spotted sandpipers, black-necked stilts, and willets was used (i.e., 0.126 kg). Because the feeding guild encompasses species with widely varying body weights, the calculation of the standard deviation of the mean body weight would have yielded an unduly wide distribution. Instead, we adopted a coefficient of variability (CV) of 15%, which is typical of body weight data for birds. The application of the adopted CV yielded a standard deviation of 0.018.

We also repeated the Monte Carlo analysis with a mean body weight of 0.044 kg (standard deviation equal to 0.0066). This body weight is representative of the smallest bird in the guild, the spotted sandpiper. Small birds tend to have higher metabolic rates and, as a result, may be at higher risk of exposure.

Body weights were assumed to be distributed normally. The entire proportion of uncertainty in this variable is likely due to variability, with little incertitude. Thus, probability bounds were not established for this input variable.

Diet

Birds belonging to this feeding guild generally forage on sandbars, mud flats, and along tidal creeks and pannes of salt marshes for brine flies, brine shrimp, crayfish, snails, tadpoles, molluscs, marine worms, and small fish (Ehrlich *et al.* 1988; Gough *et al.* 1998). Because invertebrates constitute the main food item of sediment-probing birds (Oring *et al.* 1986), our exposure model was set to consider only that prey group. Other dietary items were considered of minor importance and were omitted from our analysis.

Free Metabolic Rate (FMR)

To estimate free metabolic rate, the allometric equation derived by Nagy (1987) was used:

$$FMR = a \cdot BW(g)^b \qquad \textbf{EQUATION \#2}$$

A probabilistic approach was used to estimate *FMR* in both the Monte Carlo and probability bounds analyses, wherein distributions were derived for each of the input variables (body weight [BW, a, b]) and combined according to the above equation. The slope (a) and power term (b) distributions were based on the error statistics reported in Nagy (1987), assuming an underlying normal distribution for each. For piscivorus birds, log a had a reported mean of 0.681 and a standard error of 0.102, and b had a reported mean of 0.749 and a standard error of 0.037 (Nagy 1987). The BW distribution was described above.

Gross Energy of Invertebrates (GE_i)

Gross energies of shrimp, isopods, and crabs, which are dietary food items consumed by sediment-probing birds, were available from the literature. The gross energies of these organisms were reported as follows: shrimp = 1,100 Kcal/kg (standard deviation = 240; Thayer *et al*. 1973), isopods/amphipods = 1,100 Kcal/kg (Jorgensen *et al*. 1991), and crabs = 1,000 Kcal/kg (standard deviation = 210; Thayer *et al.* 1973). For benthic invertebrates consumed by sediment-probing birds, the mean gross energy was set to 1,070 Kcal/kg (standard deviation = 220) in the Monte Carlo analysis. The distribution for this variable was assumed to be lognormal. Incertitude was considered low for this input variable because: (1) sufficient experimental data were available to confidently estimate the mean and standard deviation, (2) the variable is easily measured and thus measurement error is low, and (3) there appears to be little difference in the gross energies of different invertebrate species. Therefore, probability bounds were not derived for this variable.

Assimilation Efficiency of Invertebrates (AE_i)

Assimilation efficiencies of waterfowl consuming aquatic invertebrates were studied by Karasov (1990). That study indicated a mean assimilation efficiency of 77% with a standard deviation of 8.4. A beta distribution was assumed for this variable with the following parameterization: alpha = 20, beta = 6.5, and scale = 1.0. This parameterization results in a distribution that has a mean close to 77%. With this distribution, there is approximately a 95% probability that assimilation efficiency will be between 58 and 90%, which would be expected given a standard deviation of 8.4 and a slightly left-skewed distribution. As with gross energy, assimilation efficiency is easily measured and several studies indicated that efficiencies vary little between different bird species consuming invertebrates (USEPA 1993). Therefore, probability bounds were not derived for this variable.

Concentration of COCs in Invertebrates (C_i)

Mercury

Concentrations of methylmercury in invertebrates were obtained from field-collected shrimp and crabs (Groups 1B and 2A) as well as sediments collected from areas of interest. Data for BI AOC included 96 observations with a mean concentration calculated at 0.0041 mg/kg and a standard deviation of 0.0517. Mean mercury levels in invertebrates were 0.031 mg/kg (SD=0.0295, n=39) for MCR AOC, 0.0231 mg/kg (SD=0.0364, n=48) for UCR AOC, and 0.0091 mg/kg (SD=0.0151, n=38) for the reference areas.

Because sediment-probing birds are likely to spatially average their exposures over extended feeding periods, we used a bootstrapping approach to estimate mean daily residues in invertebrates over 160 feeding days. The bootstrapping included 160 outer loops (days) and 1,000 inner loops (number of Monte Carlo samples). The resulting means and standard deviations are: 0.0037 mg/kg and 0.0007 for BI AOC, 0.031 mg/kg and 0.001 for MCR AOC, 0.0221 mg/kg and 0.0006 for UCR AOC, and 0.0091 mg/kg and 0.0003 for the reference areas. The variables were assumed to be lognormally distributed.

For the probability bounds analyses, the Land statistic was used to determine the lower and upper confidence limits on the mean. The resulting values were used to parameterize lognormal distributions for the AOCs and the reference areas. The values are listed in Table H1-3.

Lead

Concentrations of lead in invertebrates were obtained from field-collected shrimp and crabs (Groups 1B and 2A) as well as sediments collected from areas of interest. Data for BI AOC included 45 observations with a mean concentration calculated at 0.191

mg/kg and a standard deviation of 0.152. Mean lead levels in invertebrates were 0.097 mg/kg (SD=0.08, n=27) for MCR AOC, 0.0741 mg/kg (SD=0.445, n=26) for UCR AOC, and 0.384 mg/kg (SD=0.554, n=23) for the reference areas.

Because sediment-probing birds are likely to spatially average their exposures over extended feeding periods, we used a bootstrapping approach to estimate mean daily residues in invertebrates over 160 feeding days. The bootstrapping included 160 outer loops (days) and 1,000 inner loops (number of Monte Carlo samples). The resulting means and standard deviations are: 0.191 mg/kg and 0.005 for BI AOC, 0.097 mg/kg and 0.002 for MCR AOC, 0.068 mg/kg and 0.006 for UCR AOC, and 0.384 mg/kg and 0.384 for the reference areas. The variables were assumed to be lognormally distributed.

For the probability bounds analyses, the Land statistic was used to determine the lower and upper confidence limits on the mean. The resulting values were used to parameterize lognormal distributions for the AOCs and the reference areas. The values are listed in Table H1-3.

Selenium

Concentrations of selenium in invertebrates were obtained from field-collected shrimp and crabs (Groups 1B and 2A) as well as sediments collected from areas of interest. Data for BI AOC included 45 observations with a mean concentration calculated at 0.324 mg/kg and a standard deviation of 0.184. Mean mercury levels in invertebrates were 0.55 mg/kg (SD=0.352, n=27) for MCR AOC, 0.492 mg/kg (SD=0.253, n=45) for UCR AOC, and 0.355 mg/kg (SD=0.215, n=23) for the reference areas.

Because sediment-probing birds are likely to spatially average their exposures over extended feeding periods, we used a bootstrapping approach to estimate mean daily residues in invertebrates over 160 feeding days. The bootstrapping included 160 outer loops (days) and 1,000 inner loops (number of Monte Carlo samples). The resulting means and standard deviations are: 0.324 mg/kg and 0.002 for BI AOC, 0.55 mg/kg and 0.011 for MCR AOC, 0.492 mg/kg and 0.007 for UCR AOC, and 0.355 mg/kg and 0.006 for the reference areas. The variables were assumed to be lognormally distributed.

For the probability bounds analyses, the Land statistic was used to determine the lower and upper confidence limits on the mean. The resulting values were used to parameterize lognormal distributions for the AOCs and the reference areas. The values are listed in Table H1-3.

TCDD-TEQs

Concentrations of TCDD-TEQs in invertebrates were obtained from field-collected shrimp and crabs (Groups 1B and 2A) as well as sediments collected from areas of interest. Data for BI AOC included 11 observations with a mean concentration calculated at 22.9 ng/kg and a standard deviation of 81.5. Mean mercury levels in invertebrates were 13.9 mg/kg (SD=19.5, n=4) for UCR AOC, and 4.53 mg/kg (SD=3.66, n=3) for the reference areas. No data were available for MCR AOC.

Because sediment-probing birds are likely to spatially average their exposures over extended feeding periods, we used a bootstrapping approach to estimate mean daily residues in invertebrates over 160 feeding days. The bootstrapping included 160 outer loops (days) and 1,000 inner loops (number of Monte Carlo samples). The resulting means and standard deviations are: 22.9 mg/kg and 0.0028 for BI AOC, 13.9 mg/kg and 0.576 for UCR AOC, and 4.53 mg/kg and 0.126 for the reference areas. The variables were assumed to be lognormally distributed.

For the probability bounds analyses, the Land statistic was used to determine the lower and upper confidence limits on the mean. The resulting values were used to parameterize lognormal distributions for the AOCs and the reference areas. The values are listed in Table H1-3.

Proportion of Sediments Ingested Relative to Diet (P_s)

Incidental ingestion of contaminated sediments is an important route of uptake of contaminants for sediment-probing birds. Feeding studies on four species of sandpipers revealed that relative to diet the average intake of inorganic matter was 18% (standard deviation = 9.3; Beyer *et al.* 1994). For the Monte Carlo analysis, we assumed a beta distribution with the following parameterization: alpha = 2.0, beta = 7.0, and scale = 0.8. This parameterization results in a distribution that has a mean of 18%. With this distribution, there is close to a 95% probability that P_s will be between 6 and 42%, which would be expected given a standard deviation of 9.3, and an underlying right-skewed distribution. This variable has been measured in several studies directly relevant to sediment-probing birds (USEPA 1993). Therefore, probability bounds were not derived for this variable.

Concentration of COCs in Sediment (C_s)

Mercury

Sediment concentrations of mercury (methylmercury) in BI AOC had a mean of 0.00315 mg/kg (standard deviation = 0.00534, n=31). Mean sediment levels in MCR AOC were 0.00106 mg/kg (SD=0.00489, n=15). The UCR AOC mean sediment level was 0.00052 mg/kg (SD=0.00076, n=35), and mean reference areas sediment level was 0.00055 (SD=0.00051; n=15).

As with dietary exposure, sediment-probing birds are likely to spatially and temporally average their sediment exposure over the long term. Therefore, bootstrapping was applied to sediment residue data to derive mean sediment residues based on average daily exposure over 160 foraging days. The bootstrap means and standard deviations for each area of interest were as follows: 0.0032 mg/kg and 0.00018 for BI AOC, 0.0011 mg/kg and 0.00031 for MCR AOC, 0.00052 mg/kg and 0.00003 for UCR AOC, and 0.00055 mg/kg and 0.00002 for the reference areas. These variables were assumed to have a lognormal distribution.

For the probability bounds analyses, the Land statistic was used to determine the lower and upper confidence limits on the mean. The resulting values were used to parameterize lognormal distributions for the AOCs and the reference areas. The values are listed in Table H1-3.

Lead

Sediment concentrations of lead in BI AOC had a mean of 40.8 mg/kg (standard deviation = 93.5, n=302). Mean sediment levels in MCR AOC were 20.22 mg/kg (SD=94.3, n=157). The UCR AOC mean sediment level was 19 mg/kg (SD=48.7, n=188), and mean reference areas sediment level was 18.1 (SD=6.1; n=18).

As with dietary exposure, sediment-probing birds are likely to spatially and temporally average their sediment exposure over the long term. Therefore, bootstrapping was applied to sediment residue data to derive mean sediment residues based on average daily exposure over 160 foraging days. The bootstrap means and standard deviations for each area of interest were as follows: 40.8 mg/kg and 3.18 for BI AOC, 20.2 mg/kg and 1.91 for MCR AOC, 19 mg/kg and 1.41 for UCR AOC, and 18.1 mg/kg and 0.184 for the reference areas. These variables were assumed to have a lognormal distribution.

For the probability bounds analyses, the Land statistic was used to determine the lower and upper confidence limits on the mean. The resulting values were used to parameterize lognormal distributions for the AOCs and the reference areas. The values are listed in Table H1-3.

Selenium

Sediment concentrations of selenium in BI AOC had a mean of 0.876 mg/kg (standard deviation = 0.575, n=245). Mean sediment levels in MCR AOC were 0.62 mg/kg (SD=0.574, n=138). The UCR AOC mean sediment level was 0.562 mg/kg (SD=0.321, n=122), and mean reference areas sediment level was 0.715 (SD=0.058; n=3).

As with dietary exposure, sediment-probing birds are likely to spatially and temporally average their sediment exposure over the long term. Therefore, bootstrapping was applied to sediment residue data to derive mean sediment residues based on average daily exposure over 160 foraging days. The bootstrap means and standard deviations for each area of interest were as follows: 0.876 mg/kg and 0.017 for BI AOC, 0.62 mg/kg and 0.019 for MCR AOC, 0.562 mg/kg and 0.011 for UCR AOC, and 0.715 mg/kg and 0.002 for the reference areas. These variables were assumed to have a lognormal distribution.

For the probability bounds analyses, the Land statistic was used to determine the lower and upper confidence limits on the mean. The resulting values were used to parameterize lognormal distributions for the AOCs and the reference areas. The values are listed in Table H1-3.

TCDD-TEQs

No toxic equivalency factors (TEFs) were available for dioxin equivalents ingested by sediment-probing birds via sediment. Therefore, no sediment TCDD-TEQs were derived. The exposure calculation treated this variable as zero.

2.2.1.3 Monte Carlo Analysis

The Monte Carlo analyses for exposure combined the input distributions as specified in Equation 1. Each analysis included 10,000 trials and Latin Hypercube Sampling to ensure adequate sampling from all portions of the input distributions. The analyses were done in Crystal Ball 2000 (Decisioneering 2000). Considering all possible pairwise combinations of input variables, no dependencies were expected except for concentrations of COCs in sediments and benthic invertebrates. Therefore, no correlations were included in the Monte Carlo analyses. The incertitude associated with the nature of the dependency between concentration of COCs in sediments and benthic invertebrates will be investigated in the probability bounds analyses described below. The Monte Carlo analyses made no distinction in the way incertitude and variability were propagated; they were simply combined. We address incertitude and variability separately in the probability bounds analyses described below.

2.2.1.4 Probability Bounds Analysis

For the probability bounds analyses, we used the same table describing the state of knowledge of the input variables as was used for the Monte Carlo analyses (Table H1-2), with two exceptions.

The first exception was in regards to the incertitude about the nature of the dependency between concentrations of COCs in sediments (C_s) and benthic invertebrates (C_i). The data did not indicate an obvious relationship between these two variables. Intuitively, it would seem that the dependency between C_i and C_s would likely be positive, although we are unable to assign an exact correlation coefficient. At present, however, this partial state of knowledge regarding the nature of the dependency between C_s and C_i cannot be incorporated in the probability bounds analysis. The next best alternative given the state of development for this technique is to assume that nothing is known about the relationship between C_s and C_i. This is what was done for our analyses. The other exception was in regard to the incertitude arising from small sample size for the concentration variables. The Land statistic was used to develop distributions for mean concentration that reflect our inability to precisely specify the mean because of small sample size.

The remaining input variables were the same as used in the Monte Carlo analyses, reflecting the reasonably good state of empirical knowledge for these variables. The probability bounds analyses were run using RiskCalc, version 3.0 (Ferson *et al*. 2002) and the input is specified in Table H1-3.

2.2.2 Effects Assessment

This purpose of this section is to: (1) briefly review the literature on the effects of dietary contaminants of interest to sediment-probing birds, and (2) select the appropriate effects metrics that will be used with the results of the exposure assessment to estimate risk. We will focus on ecologically-relevant effects such as survival, reproduction, and growth. Examples of sediment-probing bird species considered in this section include sandpipers, willets, spoonbills, stilts, ibis, and dabbling ducks. Because the available toxicological information for these species is limited, the literature available for other species of birds and the results of these studies will be discussed where appropriate. Other information on the toxicity of contaminants of concern to wildlife can be found in Appendix 5 of the problem formulation document (MacDonald *et al.* 2001).

Effects data can be characterized and summarized in a variety of ways ranging from benchmarks designed to be protective of most or all species to dose-response curves for the receptor group of interest (e.g., sediment-probing birds). In this assessment, effects characterization preferentially relies on dose-response curves, but defaults to benchmarks or other estimates of effect (e.g., no observed adverse effect level (NOAEL), lowest observed adverse effect level (LOAEL) when insufficient data are available to derive dose-response curves. Effects associated with growth, survival, and/or reproduction are generally the preferred measures of effect.

The following is the hierarchy of decision criteria used to characterize effects for each COC:

1. Had bioassays with five or more treatments been conducted on the receptor group of interest or a reasonable surrogate? If yes, we estimated the dose-response relationship using the Generalized Linear Model (GLiM) framework described in Kerr and Meador (1996) and Bailer and Oris (1997). The GLiM framework involves conducting linear regression analysis on dose-response data that have been transformed to linearize the relationship (e.g., probit transformation for survival data). If not, we proceeded to 2.

2. Were multiple bioassays available that, when combined, had five or more treatments on the receptor group of interest or a reasonable surrogate? Such

bioassays would be expected to have had similar protocols, exposure scenarios and effects metrics. If yes, we estimated the dose-response relationship as in 1. If not, we proceeded to 3.

3. Had bioassays been conducted on eight or more species using similar exposure scenarios and effects metrics? If yes, we used the species sensitivity distribution (SSD) approach to estimate the doses that would cause toxicity to the 5^{th}, 50^{th} and 95^{th} percentile species on the SSD. Essentially, the SSD approach allows one to estimate risk to sediment-probing birds that have high, intermediate, and low sensitivity to COC exposure. If not, we proceeded to 4.

4. Were sufficient data available from field studies and monitoring programs to estimate concentrations or doses of COCs consistently associated with no adverse effects and with adverse effects to sediment-probing birds? If yes, we developed field-based no effects and effects measures. This approach is analogous to the approach used to develop sediment-quality guidelines for the protection of aquatic life (see Long *et al.* 1995; MacDonald *et al.* 1996; MacDonald *et al.* 2000). If not, we proceeded to 5.

5. We derived a range within which the threshold for the receptor group of interest was expected to occur. Because information on the sensitivity of the receptor of interest was lacking, it was difficult to derive a threshold that was neither biased high or low. If bioassay data are available for several other species, however, one could calculate a threshold for each to determine a threshold range that spanned sensitive and tolerant species. That range was assumed to include the threshold for the receptor group of interest.

2.2.2.1 Mercury

Methylmercury is a strong nervous system toxicant. Its ability to cross the blood brain barrier results in brain lesions, damage to the central nervous system, and spinal cord degeneration (Wolfe*et al*. 1998). Methylmercury is absorbed into the bloodstream and transported to tissues and organs throughout the body (USEPA 1997b). As a result, neurological disorders, damage to organs, and effects on growth and development are characteristic effects of MeHg. Clinical symptoms of acute poisoning include ataxia, tremors, weakness in legs and wings, muscular incoordination, paralysis, recumbency, and convulsions (USEPA 1997b; Wolfe *et al*. 1998; Eisler 2001). Adverse effects also occur from chronic exposure to low concentrations of MeHg.

The reliance of piscivorus birds on fish makes them particularly susceptible to the adverse effects of MeHg toxicity. The proportion of mercury as MeHg in fish tissues is generally greater than 90%; and increases with fish length, weight, and age (Eisler 2001). Concentration data are in wet weight (ww), unless noted otherwise.

Survival

Studies by Spalding *et al*. (2000a; 2000b) and Bouton *et al.* (1999) found behavioral abnormalities and neurologic disturbances in great egrets (*Ardea albus*) dosed with 0.5 or 5 mg MeHgCl/kg. Birds had dingy feathers, avoided sun, and were less motivated to hunt. Great egrets in the high dose group experienced severe ataxia, as well as hematologic and histologic changes. Scheuhammer (1988) observed signs of mercury poisoning in Zebra finches (*Poephila guttata*) fed 5 mg MeHg/kg dry weight (dw). On day 40 of the 77 d study, some finches began exhibiting behavioral abnormalities. Symptoms included lethargy, fluffed feathers, and difficulty flying. The first death occurred on day 68 and by day 77 four of eight finches in the dose group had died with the rest showing neurological signs of mercury poisoning (Scheuhammer 1988). Neurological signs of poisoning did not appear until mercury concentrations were $15 mg/kg in the brain and between 30 and 40 mg/kg in the liver and kidneys. The high metabolism of the Zebra finch forces it to consume a greater amount of food, and as a result, mercury. This characteristic is similar to the belted kingfisher, which has a high food intake rate on a body weight basis (USEPA 1997b).

Neurological effects and death resulted from dietary treatments of 7.2 and 10 mg/kg of MeHg dicyandiamide fed to red-tailed hawks (*Buteo jamaicensis*; Fimreite and Karstad 1971). Birds that died showed symptoms similar to those described earlier: muscular weakness, in-coordination, weight loss. Liver mercury residues ranged from 17 to 20 mg/kg in chicks that died. Lesions of axons and myelin sheaths were found in all hawks fed 7.2 and 10 mg/kg (Fimreite and Karstad 1971). Hill and Soares (1984) found similar effects in coturnix (*Coturnix japonica*) at a diet of 8 mg MeHgCl/kg over a 9 wk period. During week 8, one male and three female birds began losing muscular control. One female died during week 9 after displaying signs of severe mercury toxicity. No clinical signs were shown over the 9 wk study in coturnix fed diets containing 0.125 or 4 mg MeHgCl/kg (Hill and Soares 1984). Hill and Soares (1984) established a single oral dose LD_{50} of 18 mg/kg BW and an LC_{50} of 47 mg/kg.

Borg *et al*. (1970) fed juvenile goshawks (*Accipiter g. gentilis l.*) liver and muscle from MeHg contaminated chickens. The MeHg concentration in chicken muscle was 10 mg/kg and in liver 40 mg/kg. Diets contained either contaminated muscle and liver for a concentration of 13 mg/kg or muscle only for an average concentration of 10 mg/kg. The estimated intake of MeHg by goshawks was 0.7-1.2 mg/kg/day. Symptoms of MeHg poisoning were observed after a 2 week latency period and included inappetence, muscular weakness, ataxia, and loss of body weight.

Goshawks in the 13 mg/kg group died at 30, 38, and 47 d. One bird in the 10 mg/kg group died on day 39. MeHg concentrations in the brains of these birds ranged from 26 to 46 mg/kg and in the livers from 96 to 138 mg/kg (Borg *et al*. 1970). Concentrations similar to those used by Borg *et al*. (1970) also proved fatal to pheasants (Spann *et al*. 1972). In their study, pheasants fed 30 mg/kg ethyl mercury p-toluene, equivalent to 12.5 mg Hg/kg, died between 57 and 102 days of feeding (Spann *et al.*1972). Symptoms leading up to death were similar to previously mentioned studies.

Reproduction

Mercury's potent embryo toxicity makes reproduction one of the most sensitive endpoints. Mercury concentrations well below those required to cause effects in adults can negatively impact reproduction and survival of young (Scheuhammer 1988; USEPA 1997b). Adverse effects of mercury on reproduction include reduced hatchability caused by increased mortality of embryos, smaller clutch sizes, a greater number of eggs laid outside the nest, and aberrant behavior (USEPA 1997b; Wolfe *et al.* 1998; Eisler 2001). Reproductive effects also extend to juvenile survival (Wolfe *et al*. 1998).

Adult mallard ducks (*Anas platyrhynchos*) were unaffected by mercury concentrations of 0.5 and 3 mg MeHg/kg dw, however, effects on reproduction were evident (Heinz 1979). Dosed hens laid more eggs outside the nestbox than controls, laid fewer sound eggs, and had less ducklings survive past one week (Heinz 1979). Duckling behavior was also affected. Treated ducklings had longer response times to tape recorded maternal calls than controls (Heinz 1979). Reproduction was similarly effected in black ducks (*Anas rubripes*) fed 3 mg MeHg dicyandiamide/kg over two reproductive seasons (Finley and Stendell 1978). The most harmful effects were on hatchability and duckling survival. Other effects were observed for clutch size, egg production, and the number of eggs incubated. Mercury concentrations in whole embryos that failed to hatch averaged 9.62 and 6.08 mg/kg for the first and second year, respectively. Brain mercury concentrations in dead ducklings ranged from 3.25 to 6.98 mg/kg and displayed lesions associated with mercury poisoning (Finley and Stendell 1978). Fimreite (1971) found comparable effects in pheasants fed 2-3 mg MeHg dicyandiamide/kg for 12 weeks. The number of shell-less eggs increased and egg weights decreased, as did hatchability and the number of fertilized eggs. Mercury concentrations in unhatched eggs ranged from 0.5 to 1.5 mg/kg (Fimreite 1971). A dietary dose of 10 mg/kg of ethyl mercury *p*-toluene (mercury equivalent of 4.2 mg/kg) reduced egg production 50-80% and increased mortality in eggs that were laid (Spann *et al.* 1972). A sample of treated eggs had an average mercury concentration of 1.5 mg/kg, and a range of 0.3 to 3.1 mg/kg (Spann *et al*. 1972).

Field Surveys

Common loons (*Gavia immer*) in northwestern Ontario displayed reduced nest site fidelity and laid fewer eggs in areas where mercury concentrations in prey averaged >0.4 mg/kg. Adult loon brain concentrations of mercury between 2 and 3 mg/kg were also associated with adverse effects on reproductive behavior (Barr 1996). Monteiro and Furness (2001) observed no clinical signs of poisoning in a single oral dose experiment with MeHg on free-living Cory's Shearwater chicks (*Calonectris dimoedea*). Exposure levels ranged from 0.9 to 2.5 mg/kg body weight and the researchers noted that they were similar to the no adverse effects level (NOAEL) of 2.5 mg/kg found by Scheuhammer (1988). Monteiro and Furness (2001) estimated that the highest average mercury brain concentration in the experiment was 3.4 mg/kg. This concentration was based on a blood:brain ratio of 0.78 calculated from adult Cory's shearwaters concentrations.

Effects Metrics

The most sensitive responses in birds exposed to methylmercury are associated with reproduction following long-term exposures. None of the available reproduction studies included species that could be considered piscivorus birds or reasonable surrogates. Mallards are often used in laboratory studies, however, their foraging behaviour is considerably different from the hypothetical piscivorus receptor. The hypothetical piscivorus receptor bears much more of a resemblance to birds such as brown pelicans, osprey, belted kingfishers and terns. Its diet primarily consists of fish and lesser amounts of invertebrates. The lack of toxicity data for piscivorus birds exposed to methylmercury precludes the development of a dose-response curve (using either a single or multiple studies) or the derivation of a NOAEL and LOAEL for piscivorus birds (*i.e.*, Options 1-3 in the hierarchy of decision criteria for choosing effects metrics are unavailable).

No field data were available to develop field-based benchmarks for piscivorus birds, which eliminates Option 4 for choosing effects metrics.

The final option for choosing an effects metric for piscivorus birds exposed to methylmercury is to derive a range within which the threshold for this receptor group is expected to occur. The most sensitive reproductive response was observed in mallard ducks exposed to methylmercury for three generations (Heinz 1974; 1979; Heinz and Locke 1975). In this study, 0.5 mg Hg/kg (as methylmercury dicyanamide) led to small, but significant reductions in clutch size and duckling survival. Similarly, Fimreite (1971) estimated the threshold egg concentration for hatchability to be between 0.5 and 1.5 mg Hg/kg for ring-necked pheasants. It would therefore seem reasonable to select 0.5 mg Hg/kg in the diet as the lower bound of the threshold range for piscivorus birds exposed to methylmercury. This dietary concentration was multiplied by the food intake rate of mallard ducks (0.128 kg/day, as measured by

Heinz 1979) and normalized to their body weight (1 kg, Heinz *et al*. 1989) to derive the corresponding dose:

$$LT = \left(\frac{0.500\, mg\ Hg}{kg\ diet} \times \frac{128\ g\ food}{day} \times \frac{1.00\ kg}{1000\ g} \right) / \, 1.00\ kg\ BW$$
$$= 0.0640\, mg \,/\, kg\, bw \,/\, day$$

EQUATION #3

where *LT* is the lower threshold dose and *BW* is body weight.

Survival and reproductive data on effects reveal a broad range of bird taxa that are severely affected by dietary concentrations of methylmercury / 10 mg/kg None of the tested species, which included mallards (Heinz and Hoffman 1998), goshawks (Borg *et al*. 1970), ring-necked pheasants (Spann *et al*. 1972), white leghorn chickens (Scott 1977) and Japanese quail (Hill and Soares 1984; Scott 1977), were able to tolerate dietary concentrations of methylmercury close to or greater than 10 mg/kg. The highest level of methylmercury in the diet that did not cause adverse impacts to a test species was 6 mg/kg. The test species was red-tailed hawks (Fimreite and Karstad 1971). It would therefore seem reasonable to select 6 mg Hg/kg in the diet as representing the tolerant end of the threshold range for piscivorus birds exposed to methylmercury. This dietary concentration was multiplied by the food intake rate of red-tailed hawks (0.109 kg/day, Craighead and Craighead 1969) and normalized to their body weight (1.126 kg, Dunning 1984) to derive the corresponding dose:

$$UT = \left(\frac{6.00\, mg\ Hg}{kg\ diet} \times \frac{109\ g\ food}{day} \times \frac{1.00\ kg}{1000\ g} \right) / \, 1.13\ kg\ BW$$
$$= 0.581\, mg \,/\, kg\, bw \,/\, day$$

EQUATION #4

where *UT* is the upper threshold dose and *BW* is body weight.

Therefore, the threshold range for piscivorus birds exposed to methylmercury is 0.0640 to 0.581 mg Hg/kg bw/day.

2.2.2.2 Lead

Inorganic lead is toxic to a broad range of organs and tissues due to its activity as a metabolic poison (Kendall *et al*. 1996). Lead alters the biological function of various enzymes (lead displaces essential metals from enzyme proteins) and this has the effect of producing severe degeneration in the central nervous system, blood production (inhibits iron activity), and kidney activity (Kendall *et al*. 1996). Lead also exhibits adverse effects on reproduction including premature abortions and mortality of neonates. Effects on the immune system result in increased susceptibility of exposed animals to bacterial and viral infections (Kendall *et al*. 1996). Chronic exposure to lead is often associated with loss of body weight, behavioural changes, and partial paralyses (Sanderson and Belrose 1986; Heitmeyer *et al*. 1993).

Metallic lead is toxic to birds when administered as a powder or as lead shot. Lead shot is retained in the gizzard where it is quickly ground down to lead powder. The rate of pellet erosion is very rapid. As much as 70% of the lead contained in pellets is lost within first five days while in the gizzard. Pellets disappear completely within 35 days (Cook and Trainer 1966). Some birds are very sensitive and one ingested pellet is sufficient to cause mortality (WHO 1989).

Survival

Administration of lead pellets (by gavage) at doses of 0, 70, 140, and 240 mg per bird (0, 583, 875, and 1,500 mg/kg bw or 0, 17, 26, and 44 mg/kg bw/day) to a group of 25 adult mourning doves (*Zenaida macroura*; housed outdoors in winter) resulted in mortality rates of 0, 24, 60, and 52%, respectively, during 34 days post exposure (Buerger *et al*. 1986; Marn *et al*. 1988). Authors indicated that the LD_{50} for wild doves ranged between 140 and 280 mg/bird (875 to 1,750 mg/kg bw). Repeated dosing of adult females with 70 mg /bird (583 mg/kg bw) in an indoor environment produced no mortalities. In a similar experiment, Castrale and Oster (1993) administered lead shot to adult mourning doves. Doses of 0, 70, 140, and 240 mg/bird (0, 583, 875, and 1,500 mg/kg bw or 0, 21, 31, and 54 mg/kg bw/ day) resulted in 10 to 30% mortality 4-weeks post exposure. A dose of 400 mg/bird (2,614 mg/kg bw or 290 mg/kg bw/day) administered to ringed turtle-doves (*Streptopelia risoria*; housed at 6 deg C) resulted in 71% mortality within 9 days of post exposure. However, no mortalities were observed when the experiment was repeated at room temperature (Kendall *et al*. 1981; 1982).

Gallinaceous birds such as bobwhite quail (*Colinus virginianus*) exposed to 0, 350, and 700 mg/bird (0, 3,500, and 7,000 mg/kg bw or 0, 125, 250 mg/kg bw/day) experienced mortality rates of 8, 58, 67, and 95%, respectively (Damron and Wilson 1975). The birds were exposed in batches of 12 adults fed lead pellets (once to three times per week) for 4 weeks. Juvenile birds (10-week old), exposed to 50 mg/day/bird (1,000 mg/kg bw/day) for 6 weeks experienced 10% mortality. Adult broiler chickens that received lead acetate at the dose of 200 mg/kg bw/day for 40 days exhibited 27% mortality (Brar *et al*. 1997).

Reiser and Temple (1981) administered a dose of 3 mg/kg bw/day by gavage to red-tailed hawk, rough-legged hawk, and golden eagle for 30 weeks. Of all the birds tested, four birds died during the exposure period. Bald eagles that ingested a total of 2,000 mg of lead pellets (317 mg/kg bw or 32 mg/kg bw/day) started to die within 10 days of the exposure (Pattee *et al*. 1981). One bird became blind and was euthanized at the end of the experiment. Note that ingestion of lead pellets may not accurately indicate exposure to dissolved lead, a form of lead that is associated with toxicity. Patee *et al.* (1981) estimated that only 1 to 20% of the lead pellet dose is available for toxic action. Thus, the authors suggested that the doses that the eagles were exposed to ranged from 20 to 200 mg/bird (3 to 32 mg/kg bw or 0.022 to 0.24 mg/kg bw/day).

A dietary exposure of American kestrels to powdered metallic lead at a concentration of up to 5,000 mg/kg diet for 5 days (413 mg/kg bw/day) had no effect on mortality rates during the 13-day experiment (Hill and Camardese 1986). In another experiment, American kestrels were exposed to powdered metallic lead administered in corn oil daily for 10 days. The exposure rates were 25, 125, and 625 mg/kg b w/day. The highest dose resulted in 40% mortality (4 out of 10 birds died between day 4 and day 10 of the experiment; Hoffman *et al.* 1984). Kestrels administered one #8 shot (70 mg/bird; 700 mg/kg bw) or fed continuously with lead-tainted mallard (*Anas platyrhynchos*) meat (0.034 mg/kg meat) for 60 days showed no adverse signs with respect to increased mortality (Stendell 1980).

Dieter and Finley (1978) administered a single lead pellet (200 mg lead/bird; 74 mg/kg bw) to male and female mallard ducks. This resulted in mortality of 2 out of 60 birds in the test. The dead birds showed typical signs of lead poisoning. In comparison, ring-necked ducks (*Aythya collaris*) that swallowed a single #4 lead pellet (equivalent dose of 334.8 mg/kg bw) showed 15% mortality (Mautino and Bell 1986).

Willow grouse (*Lagopus lagopus*), that received three #6 lead shot (300 mg/bird; 600 mg/kg bw or 40 mg/kg bw/day) experienced 22% mortality. The deaths occurred when the birds lost 35 to 54 percent of their body weight (8 to 15 days after dosing). Fimreite (1984) estimated that the amount of lead eroded from the pellets was between 85 and 177 mg/bird.

Sublethal Effects

White Carneaux pigeons exposed to 12, 36, or 72 mg lead/kg bw/day by gavage exhibited overt signs of toxicity at the two highest doses. The toxicity signs included stasis, motor incoordination, and severe wasting of muscles and loss of body weight (Cory *et al.* 1980). Significant losses in body weight were also reported by Castrale and Oster (1993) who exposed mourning doves to at least 140 mg lead/bird (1,170 mg/kg bw). Buerger *et al.* (1986) reported that mourning doves given one lead pellet (70 mg/bird; 583 mg/kg bw) laid eggs whose hatchability was significantly reduced. The reduced hatchability was attributed to increased mortality of embryos. A dose of 280 mg/bird (2,154 mg/kg bw) resulted in testicular atrophy and degeneration of seminiferous tubules in ringed turtle-doves (Veit *et al.* 1983). A dose of 400 mg/bird (3,076 mg/kg bw or 220 mg/kg bw/day) administered to ringed turtle-doves (*Streptopelia risoria*; housed at 20 deg C) had no effect on body weight within 14 days post treatment. It was also estimated (using a regression model) that 70% of the inserted lead shot was eroded in the gizzard during the course of the experiment (Kendall *et al*. 1981; 1982). No effects on reproduction or body weight were observed in mourning doves administered 70 mg/bird (583 mg/kg bw; Carrnigton and Mirarchi 1989). The birds were given one lead pellet and then released into the wild for 3 weeks while being monitored with radio-transmitters.

Bald eagles that received 4,000 mg/bird (635 mg/kg bw) as lead pellets showed anaemia and enzymatic changes indicative of liver and kidney failure (Hoffman *et al*.

1981). In another experiment, the researchers exposed American kestrels to 125 mg/kg in diet (10 mg/kg bw/day). Effects included anaemia and alterations in the brain, liver, and the kidney. No effects on body weight were reported by Franson *et al*. (1982) who exposed American kestrels to dietary levels of lead powder reaching 50 mg/kg diet (4.0 mg/kg bw/day). In that experiment, 1 to 6-year old male and female birds were exposed to 0, 10, and 50 mg/kg of powdered lead in diet for 7 months. Also, no gross lesions were observed at necropsy. Custer *et al*. (1984) fed American kestrels with a diet of biologically-incorporated lead for 60 days. The level of 28 mg/kg bw/day had no effect on the body weight of kestrels. Hoffman *et al*. (1985) observed that American kestrel nestlings exposed to 125 mg/kg diet (10 mg/kg bw/day) metallic lead experienced impaired growth. In another experiment, kestrels fed diet containing 0, 10, or 50 mg/kg (0, 0.83, 4.0 mg/kg bw/day) for 7 months showed no impaired survival, egg laying, fertility, or eggshell thickness (Pattee 1984). A growth NOEC was estimated at 125 mg/kg diet. American kestrels exposed to powdered metallic lead administered in corn oil daily for 10 days showed reduced growth rates. The exposure rates were 25, 125, and 625 mg/kg bw/day. The doses of 125 and 625 mg/kg bw/day reduced weight gain by 16% and 39%, respectively (Hoffman *et al*. 1984). Haegele *et al*. (1974) estimated a reproduction NOEC of 100 mg/kg for this bird. That study examined effects of dietary inorganic lead on eggshell thickness. Reiser and Temple (1981) administered a dose of 3 mg/kg/day by gavage to red-tailed hawk, rough-legged hawk, and golden eagle for 30 weeks. Eight birds exhibited depression and anorexia.

Ring-necked ducks (*Aythya collaris*) that ingested a single #4 lead pellet (equivalent dose of 334.8 mg/kg bw) showed physical signs of lead toxicity including ataxia and loss of body weight. These symptoms were apparent 10 days after dosing (Mautino and Bell 1986). Wild black ducks (*Anas rubripes*) and mallards (*Anas platyrhynchos*), exposed to 200 mg/bird as lead shot (740 mg/kg bw) experienced reduced body weights monitored over two weeks post exposure (Chasko *et al*. 1984). Reductions ranged between 20 and 30%. In another experiment, wild black ducks and mallard ducks were given one lead shot pellet (200 mg; 143 mg/kg bw). Within 14 days of exposure (spring/fall conditions), the birds had reduced body weights. The drop in body weight was small (3 to 4%), but it was significant at $p<0.01$ (Rattner *et al*. 1989). The experiment was repeated in summer and no changes in body weights between control and treated birds were observed.

Bobwhite quail (*Colinus virginianus*) exposed to 350 mg/bird (3,500 mg/kg bw or 125 mg/kg/ day) had reduced body weight (18% lower than control). The birds were exposed in batches of 12 adults fed lead pellets (once to three times per week) for 4 weeks (Damron and Wilson 1975). A reproduction NOEC was estimated at 1,500 mg/kg diet (207 mg/kg bw/day; Damron and Wilson 1975). In comparison, a reproduction NOEC for Japanese quail (*Coturnix coturnix japonica*) was estimated at 100 mg/kg diet (13.8 mg/kg bw/day; Morgan *et al*. 1975). Exposure of Japanese quail (*Coturnix coturnix japonica*) to 1.0 mg lead acetate/kg feed (0.20 mg/kg bw/day) for 12 weeks resulted in reduced egg production and exposure to 0.5 mg/kg in feed (0.1 mg/kg bw/day) was associated with delayed onset of sexual maturity (Edens 1976). Effects on growth reduction were examined in another experiment where quail were exposed to 0, 50, or 500 mg/kg diet for 7 weeks. The highest treatment level (500 mg/kg diet; 98 mg/kg bw/day) caused a 16% drop in body weight (Edens and Melvin 1989).

Adult chickens exposed to lead acetate at the dose of 200 mg/kg bw/day for 40 days exhibited lead toxicity symptoms ranging from decreased feed intake and weight loss to muscular weakness and difficulty in respiration (Brar *et al*. 1997). Bakalli *et al*. (1995) studied the effects of lead sulfate and lead acetate on broiler chickens. The exposure regime consisted of administering 0, 0.1, 0.5, or 1.0 mg/kg feed to 12-h old chicks. Body weights and feed consumption were examined at 18 and 42 days after hatching. Results showed that the exposure regime caused a linear decrease in body weight gain. However, only body weight associated with the highest dose was significantly different from controls. There was no difference in response to lead acetate and lead sulfate. Edens and Garlich (1983) spiked the feed of domestic chicken and Japanese quail with lead acetate. The level of 1.8 mg/kg bw/day for 5 weeks had the effect of decreasing egg production by 28%. The dietary dose of 26.1 mg/kg bw/day had the effect of decreasing egg production by 77% after 4 weeks of exposure. Donaldson and McGowan (1989) exposed 20-d old chickens to dietary levels of lead. The 14-day treatments included doses of 0, 500, 1,000, and 2,000 mg/kg diet (0, 44, 88, and 175 mg/kg bw/day). The authors observed a dose-related reduction in body weight. The drop in body weight was significant at the lowest dose tested (500 mg/kg diet; or 44 mg/kg bw/day). Bakalli *et al.* (1995) Donaldson and McGowan (1989) and Damron *et al*. (1969) indicate that the growth NOECs for chicken (*Gallus domesticus*) range from 1 to 1,200 mg/kg diet.

Willow grouse (*Lagopus lagopus*), that received three #6 lead shot were observed to respond by reducing food consumption, becoming emaciated, and in two out of nine cases, dying. The deaths occurred when the birds lost 35 to 54 percent of their body weight (8 to 15 days after dosing). Fimreite (1984) estimated that the amount of lead eroded from the pellets was between 85 and 177 mg/bird.

Field Surveys

No field studies relating lead concentrations to effects on sediment-probing birds were found.

Effects Metrics

The most sensitive endpoints for birds exposed to lead are those associated with reproduction and long-term exposure via the oral route. However, none of the available reproduction studies include species that are sediment-probing or are a reasonable surrogate. As mentioned previously, although mallards and other ducks forage in or near sediments, they bear little physical resemblance to the hypothetical sediment-probing bird developed for this exposure assessment. Our hypothetical sediment-probing bird resembles sandpipers, willets and stilts and has a much lower body weight and higher free metabolic rate than do mallards and other ducks.

The lack of toxicity data for sediment-probing birds exposed to lead precluded the development of a dose-response curve (using either a single or multiple studies) or the derivation of a NOAEL and LOAEL for sediment-probing birds (i.e., options 1-3 in the hierarchy of decision criteria for choosing effects metrics are unavailable). No field data were available to develop field-based benchmarks for sediment-probing birds, which eliminates option 4 for choosing effects metrics. The only option remaining for choosing an effects metric for sediment-probing birds exposed to lead is the derivation of a range of threshold values for impaired reproduction associated with dietary intake.

The most sensitive reproductive response was observed for Japanese quail, which was exposed to 1.0 mg/kg feed (lead acetate) for 12 weeks. This exposure regime resulted in reduced egg production (fecundity; Edens 1976). The level of 1.0 mg Pb/kg ww in the diet was chosen to represent the sensitive end of the threshold range. This dietary concentration was equivalent to daily exposure of 0.20 mg/kg bw/day.

For the calculation of the upper threshold bound we have chosen a study by Morgan *et al*. 1975 who reported a reproduction NOEC for Japanese quail (*Coturnix coturnix* *japonica*) at 100 mg/kg diet (13.8 mg/kg bw/day; Morgan *et al*. 1975). Thus, the threshold range for sediment-probing birds exposed to lead is 0.20 to 13.8 mg Pb/kg bw/day.

2.2.2.3 Selenium

Selenium is an element required by birds and wildlife for good health. However, the range in concentration from healthy to toxic levels is very narrow (Heinz 1996). In nature, birds are exposed to different forms of selenium, each with varying degrees of accumulation and toxicity. Inorganic forms of selenium, such as selenite and selenate, are toxic to birds, but not to the same extent as organic selenides. Of these, selenomethionine is considered to be the most toxic and the most likely to harm birds (Heinz 1996). Symptoms of selenium toxicity (selenosis) include decreased body weight and emaciation, hepatotoxicity, histologic lesions, and reproductive effects (Heinz 1996; Eisler 2001). The toxicity of different forms of selenium to birds has received little attention in laboratory studies and there is still much to be learned. Concentration data are in wet weight (ww), unless noted otherwise.

Survival

Laboratory studies have found that concentrations of selenium in the diet equal to or below 5 mg/kg are not associated with a decrease in bird health or survival. Mallards fed seleno-DL-methionine for 14 weeks at concentrations of 1, 2, or 4 mg/kg did not exhibit physiological effects of selenosis. In fact, mallards in the 2 and 4 mg/kg treatment groups gained weight (about 10%) compared to the controls (Hoffman *et al*. 1991). American kestrels (*Falco sparverius*) similarly did not show signs of selenosis when fed a diet containing 5 mg/kg dry weight (dw) of seleno-L-methionine for 77 days (Yamamoto *et al*. 1998).

The effects of dietary concentrations above 5 mg/kg dw of selenium vary. Hoffman *et al*. (1991) reported effects associated with a diet of 8 mg/kg of seleno-DL-methionine in adult male mallards. In this 14-week study, mallards in the 8 mg/kg group exhibited symptoms of hepatotoxicity, including responses to hepatic oxidized glutathione (GSSG) concentrations and the ratio of GSSG to reduced glutathione (GSH). These findings differed from Yamamoto *et al*. (1998), who found no observable symptoms of selenosis in male and female American kestrels fed 9 mg/kg dw of seleno-L-methionine.

Adult male and female mallards fed diets containing 13.3 mg/kg of seleno-DL-methionine for 150 days did not have body weights significantly different from controls and showed no signs of selenosis (O'Toole and Raisbeck 1997). These findings were similar to those of Green and Albers (1997) for adult male mallards given 10 mg/kg of seleno-DL-methionine in the diet for 16 weeks. No histologic lesions, fat or muscle changes were identified in euthanized mallards. Heinz *et al*. (1988) described the dietary concentration of 10 mg/kg as being close to a no effects level for gross effects such as decreased food consumption, growth and survival. Their study used ducklings hatched from uncontaminated eggs and tested the effects of two types of selenium. One group of ducklings was fed a diet with 10 mg/kg of sodium selenite. Ducklings in this group had enlarged livers. The authors believed this could be attributed to the stimulation of detoxifying agents in the liver. Statistically significant reductions in food consumption and growth were also noted during the fourth and second weeks of the study, respectively. The second group of ducklings was fed 10 mg/kg dw of selenomethionine and displayed no significant effects (Heinz *et al*. 1988).

As dietary concentrations exceed those discussed above, the adverse effects of selenium toxicity become increasingly apparent and harmful. American kestrels fed a dietary concentration of 12 mg/kg seleno-DL-methionine had higher lean mass compared to total body weight and lower normalized body fat than controls or birds fed 6 mg/kg (Yamamoto and Santolo 2000). This raises concerns that wild kestrels exposed to these concentrations will be in poorer condition for the rigors of overwintering, migration, and breeding (Yamamoto and Santolo 2000).

Elevated liver and plasma GSH peroxidase was reported in mallard ducklings consuming 15 mg/kg of either seleno-L-methionine, seleno-DL-methionine, selenized yeast, or high-selenium wheat (Hoffman *et al*. 1996a). These symptoms are associated with hepatotoxicity in birds (Hoffman *et al*. 1991). This study found that seleno-L-methionine was more toxic (caused greater mortality) than seleno-DL-methionine under certain conditions. This finding is important because seleno-DL-methionine is the organic form of selenium most often used in toxicity studies. Therefore, it is possible, under some conditions, that the toxicity of selenium existing as seleno-L-methionine may be underestimated (Hoffman *et al*. 1996a). A dietary concentration of 16 mg/kg of seleno-DL-methionine, administered to adult male mallards over 14 weeks, resulted in decreased body weights and lower hemoglobin concentrations compared to controls (Hoffman *et al*. 1991).

Ducklings fed dietary treatments containing either sodium selenite or selenomethionine at 20 mg/kg consumed less food than lower dose groups. As a result, growth rates for these mallards decreased (Heinz *et al*. 1988). Lower body weights than controls was also reported by O'Toole and Raisbeck (1997) for mallards consuming 25 mg/kg of seleno-DL-methionine. Hoffman *et al*. (1996a) examined the effects of different types of selenium (seleno-L-methionine, seleno-DL-methionine, selenized yeast, and high-selenium wheat) on ducklings at dietary concentrations of 15 and 30 mg/kg. The group receiving 30 mg/kg seleno-L-methionine yielded the lowest survival rate (36%) for the 2 weeks of exposure. All ducklings had elevated levels of plasma GSH peroxidase and lowered hematocrit concentrations (Hoffman *et al*. 1996a). Thirty-two mg/kg of seleno-DL-methionine resulted in weight loss, decreased hemoglobin and hematocrit concentrations, and histopathological effects in the liver (Hoffman *et al*. 1991).

Heinz *et al*. (1988) performed a 6 week study on mallard ducklings comparing the effects of sodium selenite and selenomethionine. A dietary concentration of 40 mg/kg of sodium selenite resulted in 25% mortality. The same dietary concentration as selenomethionine resulted in 12.5% mortality. At 80 mg/kg of sodium selenite, food consumption and body weight decreased after the first week. Ninety-eight percent mortality of ducklings was observed in this group over the 6 week study. One hundred percent mortality was recorded for ducklings consuming 80 mg/kg as selenomethionine (Heinz *et al*. 1988). Heinz *et al*. (1988) observed that as the treatment level of selenium increased, so too did the amount of selenium in the liver of birds. Further, as the concentration of selenium in the liver increased, body weight decreased. Other studies have reported similar results for treatments above 30 mg/kg selenium (Heinz *et al*. 1987; Heinz *et al*. 1988; Green and Albers 1997; O'Toole and Raisbeck 1997). Albers *et al*. (1996) reported results similar to those listed above in their 16 week study of mallards exposed to either 0, 10, 20, 40, or 80 mg/kg seleno-DL-methionine. Symptoms associated with fatality observed in the 40 and 80 mg/kg groups included low body weight (25-50% below normal), emaciation, atrophy of fat and breast muscles, and liver necrosis. Symptoms observed in survivors included low body weight (10-15% below normal), poor plumage, reduced hatching success, and lipid peroxidation (Albers *et al*. 1996).

No adverse effects to survival were reported in the literature below a dietary concentration of 5 mg/kg of selenium (most often as selenomethionine). A dietary concentration of 10 mg/kg appears to be close to a no observed effects levels (Heinz *et al*. 1988). Adverse effects become more prominent as dietary concentrations exceed 10 mg/kg. Symptoms include loss of body weight, emaciation, histologic lesions, and hepatotoxicity. These symptoms become increasingly severe and harmful as dietary concentration increases, ultimately leading to death.

Reproduction

The embryo is the most sensitive avian life stage to selenium poisoning (Heinz *et al*. 1987; Hoffman and Heinz 1988; Heinz *et al*. 1989; Heinz 1996). Selenium levels in the egg provide the most sensitive measure for evaluating the potential for selenium toxicity. Effects to reproduction include embryo abnormalities, reduced hatchability, teratogenic effects, and reduced survival.

Reproductive effects occur at dietary concentrations below the threshold of adverse effects in adult birds. Heinz *et al*. (1987) studied reproductive effects from selenium using adult mallards fed either 1, 5, 10, 25, or 100 mg/kg of sodium selenite or 10 mg/kg of seleno-DL-methionine. Researchers in this study estimated that a 1000 g mallard consumes 100 g of feed per day. All 6 females and 5 of 6 males fed 100 mg/kg of sodium selenite died between days 16 and 39 of the study. None of these pairs reproduced during the study (Heinz *et al*. 1987). No effects on reproduction were found in mallards receiving 1, 5, or 10 mg/kg sodium selenite. Similarly, Heinz *et al*. (1989) found no significant difference in reproductive success between controls and mallards receiving 1, 2, or 4 mg/kg selenomethionine. Stanley *et al*. (1996) also found no difference in the reproductive success of mallards fed a diet containing 3.5 mg/kg seleno-DL-methionine.

Heinz *et al*. (1987) did not find significant difference in the fertility or proportion of eggs laid between their treatment groups of 1, 5, 10, and 25 mg/kg sodium selenite. However, hens in the 25 mg/kg group took longer to begin laying eggs and laid less frequently than the others. Laboratory studies using selenomethionine cite lower concentrations associated with adverse effects to reproduction. Mallards in the 10 mg/kg of seleno-DL-methionine group of the Heinz *et al*. (1987) study had lower hatching success (30.9%) compared to controls (65.7%) and the 10 mg/kg sodium selenite group (61.9%). Eight mg/kg selenomethione fed to female mallards accumulated to 3.5 mg/kg in the liver of hens and significantly reduced reproductive success (Heinz *et al*. 1989). Stanley *et al*. (1996) reported a decrease in reproductive success for mallards fed diets containing 7 mg/kg seleno-DL-methionine. Eggs produced at this concentration contained an average of 7.1 mg/kg selenium.

In a study on reproductive effects to black-crowned night herons, a diet containing 10 mg/kg dw seleno-DL-methionine did not significantly reduce hatching success compared to controls. Researchers did note that femur lengths of dosed birds were shorter than controls. Eggs from herons dosed at 10 mg/kg dw seleno-DL-methionine contained an average 3.3 mg/kg selenium (Smith *et al*. 1988). A study using Easter screech-owls (*Otus asio*) had results similar to Smith *et al*. (1988). A diet containing 4.4 mg/kg ww (equivalent to 10 mg/kg dw) did not significantly reduce reproduction, although femur lengths of the young were shorter than controls (Wiemeyer and Hoffman 1996).

The number of abnormal embryos was not significantly different between controls and mallards at either 1 or 5 mg/kg of sodium selenite in the diet (Heinz *et al*. 1987). Groups receiving 10 or 25 mg/kg of sodium selenite or 10 mg/kg of seleno-DL-methionine produced 11.2, 22.2, and 18.3% abnormal embryos. The effects of sodium selenite were predominantly embryotoxic, such as stunted growth, swollen necks, and fewer than normal feathers. Seleno-DL-methionine related abnormalities were mostly teratogenic, for example, bill and eye defects, twisted legs and feet, and missing toes (Heinz *et al*. 1987). Egg concentrations of selenium in the group receiving 10 mg/kg of seleno-DL-methionine averaged 4.6 mg/kg (Heinz *et al*. 1987). Similar effects to reproduction have been observed in studies using dietary concentrations of 8, 15, and 16 mg/kg selenomethionine (Heinz *et al*. 1989; Heinz and Fitzgerald 1993).

Studies frequently note reduced survival of young exposed to selenium in the weeks that follow hatching. Heinz *et al*. (1987) found the number of 21-day old ducklings produced per hen was significantly lower for birds fed 25 mg/kg of sodium selenite or 10 mg/kg of seleno-DL-methionine. Hens fed 8 mg/kg selenomethionine in diet had an average of 4.6 ducklings survive to 6 days. This compared with 8.1, 8.5, 8.2, and 7.5 for hens with 0, 1, 2, or 4 mg/kg selenomethionine (Heinz *et al*. 1989).

The dietary threshold for effects to reproduction appears to lie between 4 and 8 mg/kg. Above this threshold, effects to reproduction include decreased hatching success, increased embryotoxicity, increased teratogenic effects, and decreased survival of young birds.

Field Studies

Laboratory studies often use doses that reflect levels available in the wild. As a result, field studies on the effects of selenium provide similar insights to its toxicity as do laboratory studies. Researchers have demonstrated a high risk of embryonic deformities in birds when population liver concentrations of selenium exceed 9 mg/kg (Heinz 1996). Wild populations with mean liver concentrations of selenium below 3 mg/kg are less likely to have a significant number of deformities.

Hoffman *et al*. (2002) collected American avocet (*Recurvirostra americana*) and black-necked stilt (*Himantopus mexicanus*) eggs from three separate field sites and hatched them in a laboratory. Hatching success and malformations did not differ between the sites. The highest egg concentrations were 31.4 mg/kg dw for avocets and 20.5 mg/kg dw for black-necked stilts. These concentrations did not significantly decrease hatching success or increase the number of malformations. Avocets did, however, have decreased embryo growth and lower long bone lengths at the highest concentration. These findings are comparable with those reported for black-crowned night herons and eastern screech owls (Smith *et al*. 1988; Wiemeyer and Hoffman 1996).

Survival and reproductive impacts to aquatic wild birds have been investigated by Ohlendorf *et al*. (1986a; 1986b; 1988). Selenium concentrations in food ranged from 22 to 175 mg/kg dw. Clinical symptoms and effects to reproduction associated with selenosis were observed in aquatic birds of the area and included adult emaciation, and embryonic and duckling malformations.

Effects Metrics

The most sensitive responses in birds exposed to selenium are associated with reproduction. None of the available reproduction studies included piscivorus bird species identified as focal species for this assessment or reasonable surrogates. Many of the studies used mallards. Their foraging behaviour does not resemble the hypothetical piscivorus receptor. The hypothetical piscivorus receptor exhibits characteristics similar to brown pelican, osprey, belted kingfishers, and terns. It primarily forages on fish and lesser amounts of invertebrates. The lack of toxicity data for piscivorus birds exposed to selenium precludes the development of a dose-response curve (using either a single or multiple studies) or the derivation of a NOAEL and LOAEL for piscivorus birds (i.e., Options 1-3 in the hierarchy of decision criteria for choosing effects metrics are unavailable).

An insufficient amount of field data are available to develop field-based benchmarks for piscivorus birds of the Calcasieu Estuary, which eliminates Option 4 for choosing effects metrics.

The final option for choosing an effects metric for piscivorus birds exposed to selenium is to derive a range within which the threshold for this receptor group is expected to occur. The most sensitive reproductive response was observed in mallard ducks exposed to selenomethionine (Heinz *et al*. 1989). In this study, dietary concentrations above 4 mg selenomethione/kg resulted in significantly greater embryo malformations. Stanley *et al*. (1996) reported no reproductive effects from adult mallards dosed with 3.5 mg seleno-DL-methionine/kg. However, reduced hatching success and lower duckling weights occurred at 7 mg seleno-DL-methionine/kg. Based on these studies, 4 mg selenomethionine/kg in diet was chosen as the lower bound of the threshold range for piscivorus birds exposed to selenium. This dietary concentration was multiplied by the food intake rate of mallard ducks (0.0554 kg/day; Nagy 1987) and normalized to the body weight female mallards for the 4 mg selenomethionine dose group (1.04 kg; Heinz *et al*. 1989) to derive the corresponding dose:

$$LT = \left(\frac{4.00\,mg\ Se}{kg\ diet} \times \frac{55.4\ g\ food}{day} \times \frac{1.00\ kg}{1000\ g} \right) / \ 1.04\ kg\ BW \qquad \textbf{EQUATION \#5}$$
$$= 0.214\ mg\ /\ kg\,bw\ /\ day$$

where LT is the lower threshold dose and BW is body weight.

Studies have shown black-crowned night herons and eastern screech owls to be less sensitive to selenium than mallards (Wiemeyer and Hoffman 1996; Smith *et al*. 1988). A dietary concentration of 10 mg selenomethionine/kg dw did not significantly reduce hatching success or increase the number of malformed embryos in these species.

However, mallards fed this concentration exhibited adverse effects to reproduction (Heinz *et al*. 1987). Black-crowned night herons feed mainly on fish and invertebrates and therefore may be considered similar to the hypothetical piscivorus bird receptor. The hypothetical piscivorus bird likely has an upper threshold above mallard ducks, but within the range of the black-crowned night heron. It would seem reasonable then to select 10 mg selenomethionine/kg dw as the upper threshold of effects to piscivorus birds from selenium. This dietary concentration was multiplied by the food intake rate of black-crowned night herons (0.049 kg/day dw; calculated from Nagy 1987) and normalized to their body weight (0.883 kg; Dunning 1984) to derive the corresponding dose:

$$UT = \left(\frac{10.0\, mg\ Se}{kg\ diet} \times \frac{49.1\ g\ food}{day} \times \frac{1.00\ kg}{1000\ g} \right) / \, 0.883\, kg\ BW$$
$$= 0.556\, mg \, / \, kg\, bw \, / \, day$$

EQUATION #6

where *UT* is the upper threshold dose and *BW* is body weight.

Therefore, the threshold range for piscivorus birds exposed to selenium is 0.214 to 0.556 mg selenium/kg bw/day. The Appendix G benchmark for selenium is higher than the lower and upper toxicity thresholds that were selected for the piscivorus receptor. The benchmark was derived using higher food intake rates than those used in this assessment (Sample *et al*. 1996).

2.2.2.4 TCDD-TEQs

This section will examine the effects of TCDD (2,3,7,8-tetrachlorodibenzo-*p*-dioxin) and equivalents to piscivorus birds. The TCDD Equivalent (TEQ) approach relates the toxicity of specific PCB (polychlorinated biphenyl), PCDD (polychlorinated dibenzo-*p*-dioxin), and PCDF (polychlorinated dibenzofuran) congeners to that of TCDD. This technique provides a basis with which to compare the results of toxicity studies involving PCB, PCDD, and PCDF mixtures and congeners to the specific congener profiles of sites in the Calcasieu Estuary system and is described further in Appendix G. Literature relating to survival, growth, and reproduction was reviewed. The focal species in this section are belted kingfishers, osprey, terns, and brown pelicans. Additional species will be included in the discussion when necessary. Concentration data are in wet weight (ww), unless noted otherwise.

Survival

Nosek *et al*. (1992a) treated mature hen pheasants with single intraperitoneal TCDD injections of 6,250, 25,000, or 100,000 ng/kg bw. These birds suffered body weight loss and mortality at the two higher dose levels. All birds given the 100,000 ng/kg bw dose were dead by the sixth week and 75% of those given 25,000 ng/kg bw were dead by the twelfth week. These investigators also examined the subchronic effects of TCDD to pheasants, dosing birds weekly with 10, 100, or 1,000 ng/kg bw for ten weeks (cumulative doses of 100, 1,000, or 10,000 ng/kg bw). Fifty-seven percent of birds given the highest dose died within the 24 week experiment, while those on the lower doses experienced no mortality. Bobwhite quail, mallards and ringed turtledoves given single oral doses of TCDD were found to have 37-day LD_{50}s of 15,000, 108,000, and 810,000 ng/kg bw, respectively (Hudson *et al*. 1984). Chickens given single oral doses of 25,000 ng/kg died within 12 days (Grieg *et al*. 1973) and a 21 day oral NOAEL of 100 ng/kg/day was reported for treatments to 3 day old white leghorn chicks (Schwetz *et al*. 1973).

Reproduction

Nosek *et al*. (1992a) monitored egg production and embryonic mortality in mature hen ring-necked pheasants after weekly intraperitoneal injections of 10, 100, and

1,000 ng/kg bw TCDD. At the highest dose, egg production fell significantly compared to controls - from a cumulative total of 33 eggs per bird down to 12. Egg production was not affected at the two lower doses. Embryotoxicity significantly increased in response to the dose level. Cumulative doses of 100 and 1,000 ng/kg bw elicited insignificant increases in embryo mortality, but the 10,000 ng/kg bw dose caused 100% embryo mortality compared to 0% in controls.

The effects of egg injection of TCDD, TCDF, and PCBs have been reported by several investigators and include decreased egg production and increased embryonic mortality. Studies of egg injections with PCBs have demonstrated that when similar toxicant levels are attained in the egg via injection and via conventional maternal dietary doses, the effects to the chicks are also similar (Hoffman *et al*. 1996a; Nosek *et al*. 1993). Embryonic uptake of organochlorines from yolk is similar for substances injected into the yolk and for those accumulated naturally (Peakall and Fox 1987). Bioaccumulative environmental substances concentrate in egg yolks (Tumasonis *et al*. 1973; Custer *et al*. 1997). As a result, many studies have been conducted examining the effects of injecting environmentally relevant concentrations of PCBs into yolks. Egg yolk injected PCBs are distributed throughout the embryo, including fat tissue, liver, kidneys and bone marrow (Brunstrom *et al*. 1982). Ring-necked pheasant hens fed radiolabeled TCDD were found to eliminate approximately 1% of their body burdens into eggs, and all of the substance was deposited in the yolk, none in the albumin (Nosek *et al*. 1992b). The maternal transfer of total PCBs to eggs for several avian species was investigated by Drouillard and Norstrom (2001). Ratios of egg yolk to maternal adipose tissue PCB concentrations ranged from 0.270 in ring doves to 1.20 in chickens and pheasant.

Henshel *et al*. (1997) estimated the LD_{50} of TCDD injected into white leghorn chicken eggs yolks to be 122 ng/kg egg (by probit analysis, 146 ng/kg egg when determined by interpolation) and Powell *et al*. (1996b) observed that hatchability of white leghorn chicken eggs significantly decreased at a dose of 160 ng/kg egg TCDD injected into egg yolks. McKinney *et al*. (1976) reported that the injection of 5,000 ng/kg ww egg 2,3,7,8-TCDF (500 ng/kg ww egg TEQ) resulted in complete mortality of one-day-old white leghorn chickens within an average of 11.5 days. Chickens fed diets containing fish from a TCDD and PCB contaminated site at increasing concentrations experienced time and dose related decreases in egg hatchability (Summer *et al*. 1996). Total PCB concentrations in the diet ranged from 0.300 to 6.60 mg/kg. This corresponds to concentrations of 3.3 to 59 ng/kg diet of TCDD-TEQs, determined by the H4IIE bioassay.

Cormorant eggs were less sensitive to the effects of TCDD. Eggs collected from an isolated colony in Manitoba were injected with 4,000 ng/kg TCDD into the yolk sac. Cormorant eggs receiving 4,000 ng/kg egg TCDD suffered 50% mortality while controls experienced 28% mortality (Powell *et al*. 1997) in one experiment. The investigators then increased the dose range in a subsequent study (Powell *et al*. 1998) and observed 44.7% mortality in controls and 84.9% in eggs treated with 11,900 ng/kg egg TCDD. The LD_{50} of this second study was estimated to be 4,000 ng/kg egg.

Nosek *et al*. (1993) estimated a TCDD LD_{50} of 2,180 ng/kg ww egg to ring-necked pheasants when administered in egg yolks. Eggs were injected on day 0 of embryonic development with doses of 10, 100, 1,000, or 10,000 ng/kg egg and mortality, defined as a “failure of the hatchling to emerge completely from the shell alive”, was monitored. Eggs treated with the three lowest doses showed no significant increase in embryo mortality over controls, while the 10,000 ng/kg dose caused near total (98%) embryonic failure.

Chicken eggs that had been incubating for four days were injected with 3,3',4,4',5-pentachlorobiphenyl (PCB126) at treatment levels ranging from 0 to 2,000 ng/kg (0 to 200 ng/kg WHO TEQ; Brunstrom and Andersson 1988). After 14 days, embryonic mortality was highest in the highest treatment group (90%) compared to control groups (vehicle only = 15% embryo mortality). Brunstrom (1989) found that PCB126 was the most toxic of the congeners tested, with PCB77, 105, and 118 being 5, 1,000, and 8,000 times less toxic, respectively.

Powell *et al*. (1996a; 1996b; 1997) also investigated the embryotoxicity of PCB126 to chickens, and cormorants. Chicken eggs were yolk-injected with 100,000, 200,000, 400,000, 800,000, 1,600,000, 3,200,000, 6,400,000, and 12,800,000 ng/kg egg prior to incubation. The LD_{50} for chick embryos was estimated to be 2,300 ng/kg egg (230 ng/kg egg TEQ). Cormorant eggs were collected from Lake Winnipegosis in Manitoba, Canada and were injected with doses of PCB126 at levels of 0, 5,000, 10,000, 25,000, 50,000, 100,000, 200,000, 400,000, and 880,000 ng/kg ww egg (0 to 88,000 ng/kg ww egg TEQ). The eggs were then incubated for 21 days and candled on days 7, 14, and 21 to check for viability. Significant increases in embryo mortality were observed in the 400,000 and 880,000 ng/kg dose groups (40,000 and 88,000 ng/kg TEQ), to 87% and 100%, respectively (Powell *et al*. 1997). An LD_{50} of 158,000 ng/kg (15,800 ng/kg TEQ) was estimated. A second study involving cormorants estimated a PCB126 LD_{50} of 177,000 ng/kg ww egg (17,700 ng/kg ww egg TEQ) after a single injection into the yolk (Powell *et al*. 1998).

PCB77 is another PCB congener whose toxicity closely resembles that of TCDD. Chicken eggs injected into the yolk with 5,000 or 20,000 ng/kg egg (250 or 1,000 ng/kg egg TEQ) showed significantly higher embryonic mortality (55 and 100%) than in controls (15%). Herring gull and goose eggs injected with doses as high as 1,000,000 ng/kg egg (50,000 ng/kg egg TEQ) showed no significant increases in mortality and duck eggs showed no significant increases in mortality with doses as high as 5,000,000 ng/kg egg (250,000 ng/kg egg TEQ; Brunstrom 1988). Wild turkey embryos were also much less sensitive to PCB77 than were chickens. The Ah receptor, thought to be instrumental in the expression of TCDD and PCB toxicity, is not present in turkeys in the embryonic stage of development, and may therefore provide a basis for the species difference (Brunstrom and Lund 1988).

Henshel *et al.*(1997) compared the relative sensitivities of TCDD yolk and air sac injections into the eggs of white leghorn chickens. Eggs were injected on day 0 of embryonic development and were allowed to hatch undisturbed. The result was a significantly (60%) lower LD_{50} for the yolk route of administration. Air sac injections of PCB126 were also investigated by Hoffman *et al*. (1998) in multiple bird species. White leghorn chicken embryo was the most sensitive with an LD_{50} of 400 ng/kg ww egg (40 ng/kg ww egg TEQ), while American kestrel and common tern embryos were less sensitive with LD_{50}s of 65,000 and 104,000 ng/kg ww egg (6,500 and 10,400 ng/kg ww egg TEQ), respectively. An LD_{50} of 8,600 ng/kg egg (430 ng/kg TEQ) was calculated for chick eggs dosed with PCB77 administered into the air sac (Brunstrom and Andersson 1988).

Other Effects

Henshel (1998) dosed white leghorn chicken embryos with TCDD via yolk injection and examined the symmetry of the tectum and forebrain of the chicks' brains. Chickens suffered brain deformities, as asymmetries, at doses as low as 10 ng TCDD/kg egg administered via egg yolk injection. Herons and cormorants showed brain asymmetry at accumulated TCDD levels of 10 and 19 ng/kg egg. Investigations of the teratogenic effects of PCB126 in chicks revealed the potential for beak deformities and edema (Powell *et al*. 1996a; 1996b). Injections of PCB126 at levels of 900 ng/kg egg (90 ng/kg egg TEQ) caused a significant increase in the number of abnormal embryos per number of eggs (13/60 vs 3/59 for the vehicle control) while having no significant impact on mortality of the birds. Other abnormalities noted included small or missing eyes and curved toes.

Weight gain of chicks is also an effect of PCB and TCDD exposure. White leghorn cockerels were fed a variety of hexachlorobiphenyl congeners at 400,000,000 ng/kg diet for 21 days and body weights monitored (McKinney *et al*. 1976). Three of five congeners tested significantly inhibited weight gain of the birds, with 2,4,5,2',4',5'-HCB having the most impact (chick weight 78% of controls on day 21). One congener, 3,4,5,3',4',5'-HCB, produced 100% mortality in test animals within 11 days of the onset of the experiment. Nestling kestrels orally dosed with PCB126 to levels of 50,000, 250,000, and 1,000,000 ng/kg bw/day (5,000, 25,000 and 100,000 ng/kg bw/day TEQ) via the diet also experienced inhibited weight gain (Hoffman *et al*. 1996b). For days 4 to 10 of the study, there was a significant correlation between PCB concentration and decreased body weight. Smaller bone lengths also indicated a reduced growth rate. Humerus, radius-ulna, and tibiotarsus were all significantly shorter in the 250,000 and 1,000,000 ng/kg bw/day (25,000 and 100,000 ng/kg bw/day TEQ) test groups than controls. Embryonic exposure to PCB126 also resulted in decreased growth rates in white leghorn chickens (Powell *et al*. 1996a). Injection of 900 ng/kg egg (90 ng/kg egg TEQ) of PCB126 prior to incubation produced significantly reduced body weights by the second week and 3,000 ng/kg egg (150 ng/kg egg TEQ) of PCB77 reduced body weights compared to controls at 3 weeks (Powell *et al*. 1996a).

Field Studies

There has been some discussion in the literature regarding the relationship between adverse reproductive effects to birds observed in the field and long-lived chlorinated organic pollutants (de Voogt *et al*. 2001). PCBs and DDE are ubiquitous pollutants found at many contaminated sites. Cormorants in the Great Lakes area have a strong correlation (r^2 = 0.703) between egg mortality and bioassay-derived dioxin equivalents (Tillitt *et al*. 1992). Custer *et al*. (1999) instead suggest that DDE was primarily responsible for the observations of *in situ* decline in cormorant reproductive success in this area and that TCDD equivalents did not have a significant effect on cormorant reproductive success in Green Bay, despite significant PCB contamination. Eggs containing 299 ng TEQ/kg egg had 39% mortality while eggs from the reference site, containing only 35 ng TEQ/kg egg had 8% mortality. DDE concentrations were not included in the analyses.

Elliott *et al*. (2001) investigated the effects of organochlorine substances on the reproductive success of osprey in the Fraser and Columbia river systems. Analysis of concentrations in egg yolks and the results of laboratory incubation of eggs from the test and reference sites showed no correlation between embryonic mortality and *in ovo* substance exposure, despite hatching success ranging from 56 to 100% at the various sites. Woodford *et al*. (1998) monitored reproductive success of osprey exposed to chlorinated substances in the Wisconsin River from 1992 to 1996. Study sites included two test sites downstream of two bleached-kraft facilities and two reference sites upstream. From these sites, eggs were collected to measure contamination levels and the remaining eggs monitored for hatching and fledging rates as well as weight gain. Exposure to PCDDs, PCDFs and coplanar PCBs at these sites did not affect hatching or fledging rates, but chick growth may have been reduced at TCDD concentrations ranging from 54 to 67 ng/kg ww egg.

PCBs and dioxins have been linked to teratogenic effects in the field. Ludwig *et al*. (1996) observed a relationship between the abnormality rate (number of abnormalities per 1000 eggs) and TCDD-TEQs concentrations in double-crested cormorants and Caspian terns in the upper Great Lakes. Bill defects and edema were the most common deformities. The abnormality rate reached 14.3% in live cormorant eggs in Green Bay, WI and 28.8% in live tern eggs in Saginaw Bay, MI, both contaminated sites. The overall live cormorant egg deformity rate correlated positively with TEQs ($r^2 = 0.86$) and the overall live tern egg deformity rate did not correlate as well with TEQs ($r^2 = 0.12$).

Effects Metrics

The most sensitive responses in birds exposed to TCDD-TEQs are associated with reproduction following long-term exposure. None of the available reproduction studies included species that could be considered piscivorus birds or reasonable surrogates. The hypothetical receptor embodies characteristics similar to brown pelicans, osprey, belted kingfishers, and terns. The lack of toxicity data for piscivorus birds exposed to TCDD-TEQs precludes the development of a dose-response curve (using either a single or multiple studies) or the derivation of a NOAEL and LOAEL for piscivorus birds (i.e., Options 1-3 in the hierarchy of decision criteria for choosing effects metrics are unavailable). The field data were insufficient to develop field-based benchmarks for piscivorus birds of the Calcasieu Estuary, which eliminates Option 4 for choosing effects metrics.

The final option for choosing an effects metric for piscivorus birds exposed to TCDD-TEQs is to derive a range within which the threshold for this receptor group is expected to occur. The most sensitive response observed was reproductive success of ring-necked pheasants injected weekly with TCDD (Nosek *et al*. 1992a). In this study, 14 ng TCDD/kg bw/day did not significantly reduce reproductive success of hen pheasants, while the next highest dose of 140 ng TCDD/kg bw/day caused a decrease in cumulative egg production. This concentration will be used to represent the sensitive end of the toxicity threshold.

The upper bound of the threshold range is derived from a study on the effects of PCB126 to American kestrel hatchlings (Hoffman 1996a). In this study, the highest level of TEQ in the diet that did not cause adverse effects was 5,000 ng/kg TEQ. This dietary concentration is multiplied by the food intake rate of American kestrel hatchlings (0.00778 kg/day, Nagy 1987) and normalized to their body weight (0.076 kg, Hoffman *et al*. 1996b) to derive the corresponding dose:

$$UT = \left(\frac{5{,}000\, ng\ TEQ}{kg\ diet} \times \frac{7.78\ g\ \ food}{day} \times \frac{1.00\, kg}{1000\ g} \right) / \ 0.0760\ kg\ BW$$ **EQUATION #7**

$$= 512\ ng\ /\ kg bw\ /\ day$$

where *UT* is the upper threshold dose and *BW* is body weight.

Therefore, the threshold range for sediment-probing birds exposed to TCDD-TEQs is 14 to 512 ng TEQ/kg bw/day.

3.0 Risk Characterization

In the risk characterization phase of the probabilistic risk assessment, the results of the exposure assessment (i.e., reverse cumulative distribution functions) and effects measures are integrated to develop risk estimates. The reverse cumulative distribution function was used to identify the concentration of each COC that corresponded to probabilities of exceedance ranging from 1.0 to 0. The operation was repeated from each subreach of interest.

3.1 Results and Discussion

3.1.1 Probabilistic Exposure Assessment

Mercury – Bayou d'Inde AOC

The Monte Carlo analysis revealed that exposure of average sediment-probing birds to mercury in BI AOC could range from a minimum of 0.000211 to a maximum of 0.00457 mg/kg bw/day. The mean exposure is 0.00112 mg/kg bw/day and the median exposure is 0.00110 mg/kg bw/day. Ninety percent of exposure estimates are between 0.000557 and 0.00195 mg/kg bw/day. Figure H1-4 depicts the cumulative distribution of mercury intake rates for the hypothetical average-sized sediment probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.59], followed by gross energy of invertebrates (r_p = -0.53), the power term of FMR (r_p = 0.46), and assimilation efficiency of invertebrates of invertebrates (r_p = 0.28).

The probability bounds estimated for average-sized sediment-probing birds are depicted in Figure H1-4. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000116 and 0.664 mg/kg bw/day. The 50^{th} percentile ranges between 0.000285 and 1.34 mg/kg bw/day, and the 90^{th} percentile ranges between 0.000603 and 3.72 mg/kg bw/day. In comparison, the 10^{th} percentile of the Monte Carlo prediction is 0.000646, the 50^{th} percentile is 0.00106, and the 90^{th} percentile is 0.00169 mg/kg bw/day.

Simulations with data for the small-sized sediment-probing bird species indicated that exposure could range from a minimum of 0.000324 to a maximum of 0.00563 mg/kg bw/day. The mean exposure is 0.00146 mg/kg bw/day and the median exposure is 0.00136 mg/kg bw/day. Ninety percent of exposure estimates are between 0.000753 and 0.00246 mg/kg bw/day. Figure H1-5 depicts the cumulative distribution of mercury intake rates for the hypothetical small-sized sediment probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.62], followed by gross energy of invertebrates (r_p = -0.54), the power term of FMR (r_p = 0.38), and assimilation efficiency of invertebrates (r_p = 0.30).

The probability bounds estimated for small-sized sediment-probing birds are depicted in Figure H1-5. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000313 and 0.844 mg/kg bw/day. The 50th percentile ranges between 0.000761 and 1.69 mg/kg bw/day, and the 90th percentile ranges between 0.00161 and 4.68 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.000856, the 50th percentile is 0.00136, and the 90th percentile is 0.00218 mg/kg bw/day.

Mercury – Middle Calcasieu River AOC

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds could range from a minimum of 0.00168 to a maximum of 0.0315 mg/kg bw/day. The mean exposure is 0.00772 mg/kg bw/day and the median exposure is 0.00715 mg/kg bw/day. Ninety percent of exposure estimates are between 0.00382 and 0.0136 mg/kg bw/day. Figure H1-6 depicts the cumulative distribution of mercury intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.59], followed by gross energy of invertebrates (r_p = -0.52), the power term of FMR (r_p = 0.46), and assimilation efficiency of invertebrates (r_p = 0.29).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-6. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000444 and 0.0332 mg/kg bw/day. The 50^{th} percentile ranges between 0.000769 and 0.0526 mg/kg bw/day, and the 90^{th} percentile ranges between 0.00119 and 0.1 mg/kg bw/day. In comparison, the 10^{th} percentile of the Monte Carlo prediction is 0.00437, the 50^{th} percentile is 0.00716, and the 90^{th} percentile is 0.0117 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds could range from a minimum of 0.00551 to a maximum of 0.0371 mg/kg bw/day. The mean exposure is 0.0107 mg/kg bw/day and the median exposure is 0.01 mg/kg bw/day. Ninety percent of exposure estimates are between 0.00551 and 0.018 mg/kg bw/day. Figure H1-7 depicts the cumulative distribution of mercury intake rates for the hypothetical small-sized sediment probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.62], followed by gross energy of invertebrates (r_p = -0.54), the power term of FMR (r_p = 0.38), and assimilation efficiency of invertebrates (r_p = 0.30).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-7. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranges between 0.0006 and 0.0422 mg/kg bw/day. The 50^{th} percentile ranges between 0.00103 and 0.0667 mg/kg bw/day, and the 90^{th} percentile ranges between 0.00159 and 0.125 mg/kg bw/day. In comparison, the 10^{th}

percentile of the Monte Carlo prediction is 0.00628, the 50th percentile is 0.01, and the 90th percentile is 0.0159 mg/kg bw/day.

Mercury – Upper Calcasieu River AOC

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds could range from a minimum of 0.00114 to a maximum of 0.0219 mg/kg bw/day. The mean exposure is 0.00584 mg/kg bw/day and the median exposure is 0.00546 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0029 and 0.01 mg/kg bw/day. Figure H1-8 depicts the cumulative distribution of mercury intake rates for the hypothetical average-sized sediment probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.52), the power term of FMR (r_p = 0.44), and assimilation efficiency of invertebrates (r_p = 0.28).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-8. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000378 and 0.0254 mg/kg bw/day. The 50th percentile ranges between 0.000657 and 0.0411 mg/kg bw/day, and the 90th percentile ranges between 0.00101 and 0.08 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.00333, the 50th percentile is 0.00545, and the 90th percentile is 0.00876 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds could range from a minimum of 0.00164 to a maximum of 0.0287 mg/kg bw/day. The mean exposure is 0.00757 mg/kg bw/day and the median exposure is 0.00712 mg/kg bw/day. Ninety percent of exposure estimates are between 0.00391 and 0.0128 mg/kg bw/day. Figure H1-9 depicts the cumulative distribution of mercury intake rates for the hypothetical small-sized sediment probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.64], followed by gross energy of invertebrates (r_p = -0.55), the power term of FMR (r_p = 0.36), and assimilation efficiency of invertebrates (r_p = 0.28).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-9. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000509 and 0.0323 mg/kg bw/day. The 50th percentile ranges between 0.000877 and 0.0522 mg/kg bw/day, and the 90th percentile ranges between 0.00136 and 0.1 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.00451, the 50th percentile is 0.00707, and the 90th percentile is 0.0113 mg/kg bw/day.

Mercury – Reference Areas

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds could range from a minimum of 0.000530 to a maximum of 0.00835 mg/kg bw/day. The mean exposure is 0.00243 mg/kg bw/day and the median exposure is 0.00227 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0012 and 0.0042 mg/kg bw/day. Figure H1-10 depicts the cumulative distribution of mercury intake rates for the hypothetical average-sized sediment probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.53), the power term of FMR (r_p = 0.46), and assimilation efficiency of invertebrates (r_p = 0.26).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-10. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.000402 and 0.036 mg/kg bw/day. The 50th percentile ranges between 0.000261 and 0.0187 mg/kg bw/day, and the 90th percentile ranges between 0.00042 and 0.0427 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.00138, the 50th percentile is 0.00228, and the 90th percentile is 0.00366 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds could range from a minimum of 0.000746 to a maximum of 0.0108 mg/kg bw/day. The mean exposure is 0.00315 mg/kg bw/day and the median exposure is 0.00295 g/kg bw/day. Ninety percent of exposure estimates are between 0.00162 and 0.0 531 g/kg bw/day. Figure H1-11 depicts the cumulative distribution of mercury intake rates for the hypothetical small-sized sediment probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.62], followed by gross energy of invertebrates (r_p = -0.56), the power term of FMR (r_p = 0.36), and assimilation efficiency of invertebrates (r_p = 0.28).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-11. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.0002 and 0.0149 mg/kg bw/day. The 50th percentile ranges between 0.000348 and 0.0237 mg/kg bw/day, and the 90th percentile ranges between 0.000537 and 0.0452 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.00184, the 50th percentile is 0.00295, and the 90th percentile is 0.00468 mg/kg bw/day.

Lead – Bayou d'Inde AOC

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds to lead in BI AOC could range from a minimum of 0.333 to a maximum of 8.97 mg/kg bw/day. The mean exposure is 1.99 mg/kg bw/day and the median exposure is 1.85 mg/kg bw/day. Ninety percent of exposure estimates are between 0.978 and 3.46 mg/kg bw/day. Figure H1-12 depicts the cumulative distribution of lead intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.55), the power term of FMR (r_p = 0.44), and assimilation efficiency of invertebrates (r_p = 0.29).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-12. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.167 and 40.8 mg/kg bw/day. The 50th percentile ranges between 0.285 and 66.4 mg/kg bw/day, and the 90th percentile ranges between 0.444 and 126 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 1.12, the 50th percentile is 1.85, and the 90th percentile is 3.01 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds to lead in BI AOC could range from a minimum of 0.511 to a maximum of 7.89 mg/kg bw/day. The mean exposure is 2.6 mg/kg bw/day and the median exposure is 2.42 mg/kg bw/day. Ninety percent of exposure estimates are between 1.31 and 4.34 mg/kg bw/day. Figure H1-13 depicts the cumulative distribution of lead intake rates for the hypothetical small-sized sediment probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.63], followed by gross energy of invertebrates (r_p = -0.55), the power term of FMR (r_p = 0.37), and assimilation efficiency of invertebrates (r_p = 0.29).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-13. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.226 and 51.7 mg/kg bw/day. The 50th percentile ranges between 0.38 and 84.2 mg/kg bw/day, and the 90th percentile ranges between 0.6 and 158 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 1.51, the 50th percentile is 2.4, and the 90th percentile is 3.84 mg/kg bw/day.

Lead – Middle Calcasieu River AOC

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds could range from a minimum of 0.198 to a maximum of 3.54 mg/kg bw/day. The mean exposure is 0.983 mg/kg bw/day and the median exposure is 0.917 mg/kg bw/day. Ninety percent of exposure estimates are between 0.489 and 1.71 mg/kg bw/day. Figure H1-14 depicts the cumulative distribution of lead intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.53), the power term of FMR (r_p = 0.44), and assimilation efficiency of invertebrates (r_p = 0.27).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-14. The 10th percentile of the probability envelope

formed by the lower and upper bounds ranges between 0.0769 and 20.7 mg/kg bw/day. The 50^{th} percentile ranges between 0.132 and 337 mg/kg bw/day, and the 90^{th} percentile ranges between 0.205 and 64.5 mg/kg bw/day. In comparison, the 10^{th} percentile of the Monte Carlo prediction is 0.565, the 50^{th} percentile is 0.918, and the 90^{th} percentile is 1.48 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds could range from a minimum of 0.322 to a maximum of 5.64 mg/kg bw/day. The mean exposure is 1.27 mg/kg bw/day and the median exposure is 1.2 mg/kg bw/day. Ninety percent of exposure estimates are between 0.66 and 2.15 mg/kg bw/day. Figure H1-15 depicts the cumulative distribution of lead intake rates for the hypothetical small-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.63], followed by gross energy of invertebrates (r_p = -0.55), the power term of FMR (r_p = 0.37), and assimilation efficiency of invertebrates (r_p = 0.28).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-15. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranges between 0.104 and 26.3 mg/kg bw/day. The 50^{th} percentile ranges between 0.174 and 42.8 mg/kg bw/day, and the 90^{th} percentile ranges between 0.275 and 80.8 mg/kg bw/day. In comparison, the 10^{th} percentile of the Monte Carlo prediction is 0.745, the 50^{th} percentile is 1.18, and the 90^{th} percentile is 1.8 mg/kg bw/day.

Lead – Upper Calcasieu River AOC

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds could range from a minimum of 0.152 to a maximum of 3.71 mg/kg bw/day. The mean exposure is 0.92 mg/kg bw/day and the median exposure is 0.86 mg/kg bw/day. Ninety percent of exposure estimates are between 0.458 and 1.59 mg/kg bw/day. Figure H1-16 depicts the cumulative distribution of lead intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.59], followed by gross energy of invertebrates (r_p = -0.54), the power term of FMR (r_p = 0.44), and assimilation efficiency of invertebrates (r_p = 0.27).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-16. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.0704 and 19.6 mg/kg bw/day. The 50th percentile ranges between 0.121 and 31.6 mg/kg bw/day, and the 90th percentile ranges between 0.188 and 59.6 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.525, the 50th percentile is 0.858, and the 90th percentile is 1.38 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds could range from a minimum of 0.215 to a maximum of 4.77 mg/kg bw/day. The mean exposure is 1.19 mg/kg bw/day and the median exposure is 1.11 mg/kg bw/day. Ninety percent of exposure estimates are between 0.61 and 2 mg/kg bw/day. Figure H1-17 depicts the cumulative distribution of lead intake rates for the hypothetical small-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.63], followed by gross energy of invertebrates (r_p = -0.55), the power term of FMR (r_p = 0.36), and assimilation efficiency of invertebrates (r_p = 0.29).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-17. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.095 and 24.9 mg/kg bw/day. The 50th percentile ranges between 0.161 and 40 mg/kg bw/day, and the 90th percentile ranges between 0.251 and 74.5 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.7, the 50th percentile is 1.11, and the 90th percentile is 1.76 mg/kg bw/day.

Lead – Reference Areas

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds could range from a minimum of 0.233 to a maximum of 3.38 mg/kg bw/day. The mean exposure is 0.96 mg/kg bw/day and the median exposure is 0.9 mg/kg bw/day. Ninety percent of exposure estimates are between of 0.475 and 1.67 mg/kg bw/day. Figure H1-18 depicts the cumulative distribution of lead intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.59], followed by gross energy of invertebrates (r_p = -0.52), the power term of FMR (r_p = 0.46), and assimilation efficiency of invertebrates (r_p = 0.28).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-18. The 10th percentile of the probability envelope

formed by the lower and upper bounds ranges between 0.0483 and 18.5 mg/kg bw/day. The 50th percentile ranges between 0.0848 and 29.6 mg/kg bw/day, and the 90th percentile ranges between 0.131 and 57.4 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.546, the 50th percentile is 0.9, and the 90th percentile is 1.44 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds could range from a minimum of 0.3 to a maximum of 4.3 mg/kg bw/day. The mean exposure is 1.24 mg/kg bw/day and the median exposure is 1.17 mg/kg bw/day. Ninety percent of exposure estimates are between 0.64 and 2.1 mg/kg bw/day. Figure H1-19 depicts the cumulative distribution of lead intake rates for the hypothetical small-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.62], followed by gross energy of invertebrates (r_p = -0.55), the power term of FMR (r_p = 0.35), and assimilation efficiency of invertebrates (r_p = 0.30).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-19. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.0652 and 23.5 mg/kg bw/day. The 50th percentile ranges between 0.113 and 37.5 mg/kg bw/day, and the 90th percentile ranges between 0.175 and 72 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.726, the 50th percentile is 1.17, and the 90th percentile is 1.83 mg/kg bw/day.

Selenium – Bayou d'Inde AOC

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds to selenium in BI AOC could range from a minimum of 0.022 to a maximum of 0.45 mg/kg bw/day. The mean exposure is 0.111 mg/kg bw/day and the median exposure is 0.104 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0554 and 0.193 mg/kg bw/day. Figure H1-20 depicts the cumulative distribution of selenium intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.53), the power term of FMR (r_p = 0.46), and assimilation efficiency of invertebrates (r_p = 0.27).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-20. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranges between 0.00333 and 1.23 mg/kg bw/day. The 50^{th} percentile ranges between 0.00557 and 1.87 mg/kg bw/day, and the 90^{th} percentile ranges between 0.00839 and 3.38 mg/kg bw/day. In comparison, the 10^{th} percentile of the Monte Carlo prediction is 0.0638, the 50^{th} percentile is 0.104, and the 90^{th} percentile is 0.168 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds to selenium in BI AOC could range from a minimum of 0.031 to a maximum of 0.52 mg/kg bw/day. The mean exposure is 0.144 mg/kg bw/day and the median exposure is 0.135 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0739 and 0.247 mg/kg bw/day. Figure H1-21 depicts the cumulative distribution of selenium intake rates for the hypothetical small-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.64], followed by gross energy of invertebrates (r_p = -0.55), the power term of FMR (r_p = 0.37), and assimilation efficiency of invertebrates (r_p = 0.29).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-21. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.0225 and 1.57 mg/kg bw/day. The 50th percentile ranges between 0.0371 and 2.37 mg/kg bw/day, and the 90th percentile ranges between 0.0561 and 4.23 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.843, the 50th percentile is 0.135, and the 90th percentile is 0.216 mg/kg bw/day.

Selenium – Middle Calcasieu River AOC

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds could range from a minimum of 0.0405 to a maximum of 0.685 mg/kg bw/day. The mean exposure is 0.174 mg/kg bw/day and the median exposure is 0.163 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0859 and 0.305 mg/kg bw/day. Figure H1-22 depicts the cumulative distribution of selenium intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.53), the power term of FMR (r_p = 0.45), and assimilation efficiency of invertebrates (r_p = 0.29).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-22. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranges between 0.0021 and 1.17 mg/kg bw/day. The 50^{th} percentile ranges between 0.00364 and 2.04 mg/kg bw/day, and the 90^{th} percentile ranges between 0.0092 and 5.51 mg/kg bw/day. In comparison, the 10^{th} percentile of the Monte Carlo prediction is 0.0976, the 50^{th} percentile is 0.162, and the 90^{th} percentile is 0.262 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds could range from a minimum of 0.0541 to a maximum of 0.854 mg/kg bw/day. The mean exposure is 0.226 mg/kg bw/day and the median exposure is 0.211 mg/kg bw/day. Ninety percent of exposure estimates are between 0.116 and 0.386 mg/kg bw/day. Figure H1-23 depicts the cumulative distribution of selenium intake rates for the hypothetical small-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.62], followed by gross energy of invertebrates (r_p = -0.55), the power term of FMR (r_p = 0.38), and assimilation efficiency of invertebrates (r_p = 0.30).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-23. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranges between 0.0141 and 1.49 mg/kg bw/day. The 50^{th} percentile ranges between 0.0242 and 2.59 mg/kg bw/day, and the 90^{th} percentile ranges between 0.0416 and 6.92 mg/kg bw/day. In comparison, the 10^{th} percentile of the Monte Carlo prediction is 0.132, the 50^{th} percentile is 0.21, and the 90^{th} percentile is 0.335 mg/kg bw/day.

Selenium – Upper Calcasieu River AOC

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds could range from a minimum of 0.0337 to a maximum of 0.582 mg/kg bw/day. The mean exposure is 0.156 mg/kg bw/day and the median exposure is 0.146 mg/kg bw/day. Ninety percent of exposure estimates are between 0.0773 and 0.272 mg/kg bw/day. Figure H1-24 depicts the cumulative distribution of selenium intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.54), the power term of FMR (r_p = 0.45), and assimilation efficiency of invertebrates (r_p = 0.29).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-24. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.00282 and 1.13 mg/kg bw/day. The 50th percentile ranges between 0.00468 and 1.77 mg/kg bw/day, and the 90th percentile ranges between 0.00685 and 3.2 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.0883, the 50th percentile is 0.146, and the 90th percentile is 0.237 mg/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds could range from a minimum of 0.033 to a maximum of 0.719 mg/kg bw/day. The mean exposure is 0.203 mg/kg bw/day and the median exposure is 0.189 mg/kg bw/day. Ninety percent of exposure estimates are between 0.105 and 0.344 mg/kg bw/day. Figure H1-25 depicts the cumulative distribution of selenium intake rates for the hypothetical small-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.63], followed by gross energy of invertebrates (r_p = -0.57), the power term of FMR (r_p = 0.36), and assimilation efficiency of invertebrates (r_p = 0.29).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-25. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.0191 and 1.45 mg/kg bw/day. The 50th percentile ranges between 0.0312 and 2.24 mg/kg bw/day, and the 90th percentile ranges between 0.0459 and 4 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.119, the 50th percentile is 0.189, and the 90th percentile is 0.303 mg/kg bw/day.

Selenium – Reference Areas

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds could range from a minimum of 0.0274 to a maximum of 0.491 mg/kg bw/day. The mean exposure is 0.128 mg/kg bw/day and the median exposure is 0.12 mg/kg bw/day. Ninety percent of exposure estimates are between of 0.0623 and 0.221 mg/kg bw/day. Figure H1-26 depicts the cumulative distribution of selenium intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.61], followed by gross energy of invertebrates (r_p = -0.53), the power term of FMR (r_p = 0.46), and assimilation efficiency of invertebrates (r_p = 0.29).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-26. The 10th percentile of the probability envelope

formed by the lower and upper bounds ranges between 0.00965 and 0.817 mg/kg bw/day. The 50th percentile ranges between 0.0164 and 1.4 mg/kg bw/day, and the 90th percentile ranges between 0.0265 and 3.74 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.0728, the 50th percentile is 0.12, and the 90th percentile is 0.193 mg/kg bw/day.

Selenium – Reference Areas

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds could range from a minimum of 0.039 to a maximum of 0.63 mg/kg bw/day. The mean exposure is 0.165 mg/kg bw/day and the median exposure is 0.156 mg/kg bw/day. Ninety percent of exposure estimates are between 0.085 and 0.28 mg/kg bw/day. Figure H1-27 depicts the cumulative distribution of selenium intake rates for the hypothetical small-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.62], followed by gross energy of invertebrates (r_p = -0.55), the power term of FMR (r_p = 0.36), and assimilation efficiency of invertebrates (r_p = 0.31).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-27. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.013 and 1.04 mg/kg bw/day. The 50th percentile ranges between 0.0219 and 1.77 mg/kg bw/day, and the 90th percentile ranges between 0.036 and 4.7 mg/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 0.0973, the 50th percentile is 0.155, and the 90th percentile is 0.245 mg/kg bw/day.

TCDD-TEQs – Bayou d'Inde AOC

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds to TCDD-TEQs in BI AOC could range from a minimum of 1.3 to a maximum of 24.1 ng/kg bw/day. The mean exposure is 6.03 ng/kg bw/day and the median exposure is 5.6 ng/kg bw/day. Ninety percent of exposure estimates are between 3.01 and 10.5 ng/kg bw/day. Figure H1-28 depicts the cumulative distribution of TCDD-TEQ intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.53), the power term of FMR (r_p = 0.45), and assimilation efficiency of invertebrates (r_p = 0.26).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-28. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.431 and 86.6 ng/kg bw/day. The 50th percentile ranges between 1.33 and 180 ng/kg bw/day, and the 90th percentile ranges between 2.8 and 609 ng/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 3.46, the 50th percentile is 5.59, and the 90th percentile is 9.1 ng/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds could range from a minimum of 1.8 to a maximum of 31.3 ng/kg bw/day. The mean exposure is 7.8 ng/kg bw/day and the median exposure is 7.33 ng/kg bw/day. Ninety percent of exposure estimates are between 4.03 and 13.3 ng/kg bw/day. Figure H1-29 depicts the cumulative distribution of TCDD-TEQ intake rates for the hypothetical small-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.54), the power term of FMR (r_p = 0.45), and assimilation efficiency of invertebrates (r_p = 0.30).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-29. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.581 and 111 ng/kg bw/day. The 50th percentile ranges between 1.78 and 231 ng/kg bw/day, and the 90th percentile ranges between 3.75 and 773 ng/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 4.56, the 50th percentile is 7.3, and the 90th percentile is 11.5 ng/kg bw/day.

TCDD-TEQs – Upper Calcasieu River AOC

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds to TCDD-TEQs in UCR AOC could range from a minimum of 0.791 to a maximum of 14.5 ng/kg bw/day. The mean exposure is 3.67 ng/kg bw/day and the median exposure is 3.41 ng/kg bw/day. Ninety percent of exposure estimates are between 1.82 and 6.42 ng/kg bw/day. Figure H1-30 depicts the cumulative distribution of TCDD-TEQ intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.54), the power term of FMR (r_p = 0.46), and assimilation efficiency of invertebrates (r_p = 0.28).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-30. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.0937 and 22.3 ng/kg bw/day. The 50th percentile ranges between 0.385 and 60.9 ng/kg bw/day, and the 90th percentile ranges between 1.08 and 275 ng/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 2.1, the 50th percentile is 3.39, and the 90th percentile is 5.56 ng/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds could range from a minimum of 1.05 to a maximum of 19.4 ng/kg bw/day. The mean exposure is 4.75 ng/kg bw/day and the median exposure is 4.45 ng/kg bw/day. Ninety percent of exposure estimates are between of 2.44 and 8.05 ng/kg bw/day. Figure H1-31 depicts the cumulative distribution of TCDD-TEQ intake rates for the hypothetical small-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.54), the power term of FMR (r_p = 0.45), and assimilation efficiency of invertebrates (r_p = 0.30).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-31. The 10th percentile of the probability envelope formed by the lower and upper bounds ranges between 0.127 and 28.5 ng/kg bw/day. The 50th percentile ranges between 0.512 and 77.9 ng/kg bw/day, and the 90th percentile ranges between 1.45 and 348 ng/kg bw/day. In comparison, the 10th percentile of the Monte Carlo prediction is 2.76, the 50th percentile is 4.44, and the 90th percentile is 7.1 ng/kg bw/day.

TCDD-TEQs – Reference Areas

The Monte Carlo analysis revealed that exposure of average-sized sediment-probing birds could range from a minimum of 0.247 to a maximum of 5.47 ng/kg bw/day. The mean exposure is 1.2 ng/kg bw/day and the median exposure is 1.11 ng/kg bw/day. Ninety percent of exposure estimates are between of 0.584 and 2.08 ng/kg bw/day. Figure H1-32 depicts the cumulative distribution of TCDD-TEQ intake rates for the hypothetical average-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.60], followed by gross energy of invertebrates (r_p = -0.54), the power term of FMR (r_p = 0.45), and assimilation efficiency of invertebrates (r_p = 0.30).

The probability bounds estimated for hypothetical average-sized sediment-probing birds are depicted in Figure H1-32. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranges between 0.074 and 5.14 ng/kg bw/day. The 50^{th} percentile ranges between 0.125 and 8.35 ng/kg bw/day, and the 90^{th} percentile ranges between 0.205 and 15 ng/kg bw/day. In comparison, the 10^{th} percentile of the Monte Carlo prediction is 0.68, the 50^{th} percentile is 1.11, and the 90^{th} percentile is 1.82 ng/kg bw/day.

The Monte Carlo analysis revealed that exposure of small-sized sediment-probing birds could range from a minimum of 0.326 to a maximum of 5.32 ng/kg bw/day. The mean exposure is 1.55 ng/kg bw/day and the median exposure is also 1.45 ng/kg bw/day. Ninety percent of exposure estimates are between 0.794 and 2.62 ng/kg bw/day. Figure H1-33 depicts the cumulative distribution of TCDD-TEQ intake rates for the hypothetical small-sized sediment-probing bird species.

Sensitivity analysis revealed that the slope term of the free metabolic rate relationship was the most important variable [Pearson correlation coefficient (r_p) = 0.63], followed by gross energy of invertebrates (r_p = -0.55), the power term of FMR (r_p = 0.37), and assimilation efficiency of invertebrates (r_p = 0.30).

The probability bounds estimated for hypothetical small-sized sediment-probing birds are depicted in Figure H1-33. The 10^{th} percentile of the probability envelope formed by the lower and upper bounds ranges between 0.0991 and 6.76 ng/kg bw/day. The 50^{th} percentile ranges between 0.165 and 10.8 ng/kg bw/day, and the 90^{th} percentile ranges between 0.268 and 19.1 ng/kg bw/day. In comparison, the 10^{th} percentile of the Monte Carlo prediction is 0.9, the 50^{th} percentile is 1.45, and the 90^{th} percentile is 2.3 ng/kg bw/day.

3.2 Risk Assessment

For each COC-AOC risk scenario in the Calcasieu Estuary, a low, indeterminate, and high category of risk was determined for sediment-probing birds. These categories of risk were derived using the following guidance:

1. If the probability of exceeding the lower toxicity threshold is less than 20%, the risk to birds is considered low;
2. If the probability of exceeding the upper toxicity threshold is greater than 20%, the risk to birds is considered high; and,
3. All other probabilities are considered to have indeterminate risk.

Mercury

Bayou d'Inde AOC

The Monte Carlo and lower bounds prediction for total daily intake rates of mercury by average-sized birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded at this AOC. Therefore, mercury pose low risk to the survival and reproduction of average-sized sediment-probing birds in BI AOC. However, the upper probability bound suggests (given our incertitude) that there could be as much as 100% chance that the lower toxicity threshold will be exceeded and 94% chance that the upper toxicity threshold will be exceeded. Thus, there is some uncertainty about the low risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo and lower probability bounds predictions of total daily intake rates of mercury by small-sized sediment-probing birds indicates that there is no chance that the lower or upper toxicity thresholds are being exceeded. Therefore, mercury pose low risk to the survival and reproduction of small-sized sediment-probing birds. However, the upper probability bound suggests (given our incertitude) that there could be as much as 100% chance that the lower and upper toxicity thresholds will be exceeded. Thus, there is some uncertainty about the low risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark.

The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Middle Calcasieu River AOC

The Monte Carlo and lower probability bounds predictions for total daily intake rates of mercury by average-sized birds indicate that there is no chance that the lower or upper toxicity thresholds are being exceeded in this AOC. Therefore, mercury poses low risk to the survival and reproduction of average-sized sediment-probing birds in the MCR AOC. However, the upper probability bound suggests (given our incertitude) that there could be as much as a 33% probability that the lower toxicity threshold will be exceeded (Table H1-4). Thus, there is some uncertainty about the low risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo and lower probability bounds predictions for total daily intake rates of mercury by small-sized birds indicate that there is no chance that the lower or upper toxicity thresholds are being exceeded at this subarea of interest. Therefore, mercury pose low risk to the survival and reproduction of small-sized sediment-probing birds. However, the upper probability bound suggests that there could be as much as 54% chance that the lower toxicity threshold will be exceeded given our incertitude in model parameters. Thus, there is some uncertainty about the low risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Upper Calcasieu River AOC

The Monte Carlo and lower probability bounds predictions for total daily intake rates of mercury by average-sized birds indicate that there is no chance that the lower or upper toxicity thresholds are being exceeded in the AOC. Therefore, mercury poses low risk to the survival and reproduction of average-sized sediment-probing birds in the UCR AOC. However, the upper probability bound suggests that there is a 18% probability that the lower toxicity threshold will be exceeded. Thus, there is some uncertainty about the low risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo and lower probability bounds predictions for total daily intake rates of mercury by small-sized birds indicate that there is no chance that the lower or upper toxicity thresholds are being exceeded at this subarea of interest. Therefore, mercury pose low risk to the survival and reproduction of small-sized sediment-probing birds. However, the upper probability bound suggests that there could be as much as 33% chance that the lower toxicity threshold will be exceeded. Thus, there is some uncertainty about the low risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Reference Areas

The Monte Carlo and probability bounds predictions for total daily intake rates of mercury by average-sized birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded in this AOC. Thus, mercury poses low risk to the survival and reproduction of average-sized birds in the reference areas (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 43% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo and lower probability bounds predictions for total daily intake rates of mercury by average-sized birds indicate that there is no chance that the upper or lower toxicity thresholds are being exceeded at this subarea of interest. However, the upper probability bound suggests that there could be 4% probability of exceeding the lower toxicity threshold. Thus, there is some uncertainty about the low risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of mercury exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 66% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Lead

Bayou d'Inde AOC

The Monte Carlo predictions for total daily intake rates of lead by average-sized birds indicate that there is 100% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as low as 81% according to the lower probability bound (Table H1-4).

Although, Monte Carlo results suggest that it is unlikely (0% probability) that the upper toxicity threshold will be exceeded, it still could happen because the upper probability bound on our Monte Carlo prediction is well above the upper threshold (100% probability of exceedance).

Therefore, lead poses indeterminate risks to the survival and reproduction of average-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 4% probability of total daily intake of lead exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo predictions for total daily intake rates of lead by small-sized birds indicate that there is 100% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as low as 93% because of our uncertainty in model predictions (Table H1-4).

Monte Carlo results suggest that there is 0% probability that the upper toxicity threshold will be exceeded. However, it could happen with a greater frequency because the upper probability bound on our Monte Carlo prediction is well above the upper threshold (100% probability of exceedance). Therefore, lead poses indeterminate risks to the survival and reproduction of small-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 14% probability of total daily intake of lead exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Middle Calcasieu River AOC

The Monte Carlo predictions for total daily intake rates of lead by average-sized birds indicate that there is 100% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as low as 12% because of the uncertainty in model predictions (Table H1-4).

Monte Carlo results suggest that it is unlikely (0% probability) that the upper toxicity threshold will be exceeded. However, the upper probability bound on our Monte Carlo prediction suggests that there could be as much as 100% probability that the upper threshold will be exceeded. Therefore, lead poses indeterminate risks to the survival and reproduction of average-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of lead exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo predictions for total daily intake rates of lead by small-sized birds indicate that there is 100% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as low as 36% because of our uncertainty in model.

Monte Carlo results suggest that it is unlikely (0% probability) that the upper toxicity threshold will be exceeded. However, the upper probability bound on our Monte Carlo prediction suggests that there could be as much as 100% probability that the upper threshold will be exceeded. Therefore, lead poses indeterminate risks to the survival and reproduction of small-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of lead exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Upper Calcasieu River AOC

The Monte Carlo predictions for total daily intake rates of lead by average-sized birds indicate that there is nearly 100% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as low as 7% because of our uncertainty in model predictions. (Table H1-4).

Monte Carlo results suggest that it is unlikely (0% probability) that the upper toxicity threshold will be exceeded. However, the upper probability bound on our Monte Carlo prediction suggests that there could be as much as 100% probability that the upper threshold will be exceeded. Therefore, lead poses indeterminate risks to the survival and reproduction of average-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of lead exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo predictions for total daily intake rates of lead by small-sized birds indicate that there is 100% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as low as 27% because of our uncertainty in model predictions.

Monte Carlo results suggest that it is unlikely (0% probability) that the upper toxicity threshold will be exceeded. However, the upper probability bound on our Monte Carlo prediction suggests that there could be as much as 100% probability that the upper threshold will be exceeded. Therefore, lead poses indeterminate risks to the survival and reproduction of small-sized sediment-probing birds (Table H1-4), but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of lead exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Reference Areas

Interestingly, Monte Carlo predictions for total daily intake rates of lead by average-sized birds residing in reference areas indicate that there is 100% chance that the lower toxicity threshold will be exceeded. Note that this probability could be less because our uncertainty in model predictions places the lower probability bound below the lower toxicity bound (0% chance; Table H1-4).

Monte Carlo results suggest that it is unlikely (0% probability) that the upper toxicity threshold will be exceeded. However, the upper probability bound on our Monte Carlo prediction suggests that there could be as much as 100% probability that the upper threshold will be exceeded. Therefore, lead poses indeterminate risks to the survival and reproduction of average-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of lead exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Monte Carlo predictions for total daily intake rates of lead by small-sized birds residing in reference areas indicate that there is 100% chance that the lower toxicity threshold will be exceeded. Note that this probability could be as low as 4% because of our uncertainty in model predictions (0% chance; Table H1-4).

Monte Carlo results suggest that it is unlikely (0% probability) that the upper toxicity threshold will be exceeded. However, the upper probability bound on our Monte Carlo prediction suggests that there could be as much as 100% probability that the upper threshold will be exceeded. Thus, small-sized birds feeding and reproducing in the reference areas are likely to experience lead toxicity, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of lead exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Selenium

Bayou d'Inde AOC

The Monte Carlo predictions for total daily intake rates of selenium by average-sized birds indicate that there is only 3% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as little as 0% and as much as 100% because of our uncertainty in model predictions (Table H1-4).

The Monte Carlo results also suggest that there is 0% probability that the upper toxicity threshold will be exceeded. However, that probability can be much higher because the upper probability bounds on our prediction is well above the upper threshold (100% probability). Therefore, selenium poses low risks to the survival and reproduction of average-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of selenium exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo predictions for total daily intake rates of selenium by small-sized birds indicate that there is 10% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as little as 0% and as much as 100% because of our uncertainty in model predictions (Table H1-4).

Monte Carlo results suggest that there is 0% probability that the upper toxicity threshold will be exceeded. However, it could happen with a greater frequency because the upper probability bound on our Monte Carlo prediction is well above the upper threshold (100% probability of exceedance). Therefore, selenium poses low risks to the survival and reproduction of small-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of selenium exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Middle Calcasieu River AOC

The Monte Carlo predictions for total daily intake rates of selenium by average-sized birds indicate that there is 23% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as little as 0 and as much as 100% because of our uncertainty in model predictions (Table H1-4).

The Monte Carlo results also suggest that there is 0% probability that the upper toxicity threshold will be exceeded. However, that probability can be much higher because the upper probability bounds on our prediction is well above the upper threshold (100% probability). Therefore, selenium poses indeterminate risks to the survival and reproduction of average-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of selenium exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo predictions for total daily intake rates of selenium by small-sized birds indicate that there is 49% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as low as 0% and as high as 100% given the uncertainty in model predictions (Table H1-4).

Monte Carlo results suggest that there is 0% probability that the upper toxicity threshold will be exceeded. However, the upper probability bound on our Monte Carlo prediction suggests that there could be as much as 100% probability that the upper threshold will be exceeded. Therefore, selenium poses indeterminate risks to the survival and reproduction of small-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 16% probability of total daily intake of selenium exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Upper Calcasieu River AOC

The Monte Carlo predictions for total daily intake rates of selenium by average-sized birds indicate that there is 16% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as little as 0% and as much as 100% because of our uncertainty in model predictions (Table H1-4).

The Monte Carlo results also suggest that there is 0% probability that the upper toxicity threshold will be exceeded. However, that probability can be much higher because the upper probability bounds on our prediction is well above the upper threshold (100% probability). Therefore, selenium poses low risks to the survival and reproduction of average-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of selenium exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo predictions for total daily intake rates of selenium by small-sized birds indicate that there is 37% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as low as 0% and as high as 100% given the uncertainty in model predictions (Table H1-4).

Monte Carlo results suggest that there is 0% probability that the upper toxicity threshold will be exceeded. However, the upper probability bound on our Monte Carlo prediction suggests that there could be as much as 100% probability that the upper threshold will be exceeded. Therefore, selenium poses indeterminate risks to the survival and reproduction of small-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of selenium exceeding the Appendix G benchmark.

The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Reference Areas

Monte Carlo predictions for total daily intake rates of selenium by average-sized birds residing in Reference areas indicate that there is only 6% chance that the lower toxicity threshold will be exceeded. Note that this probability could be as little as 0% and as much as 100% because of our uncertainty in model predictions (0% chance; Table H1-4).

The Monte Carlo results also suggest that there is 0% probability that the upper toxicity threshold will be exceeded. However, that probability can be much higher because the upper probability bounds on our prediction is well above the upper threshold (100% probability). Therefore, selenium poses low risks to the survival and reproduction of average-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of selenium exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Monte Carlo predictions for total daily intake rates of selenium by small-sized birds residing in Reference areas indicate that there is 18% chance that the lower toxicity threshold will be exceeded. Note that this probability could be as low as 0% and as high as 100% given the uncertainty in model predictions (0% chance; Table H1-4).

Monte Carlo results suggest that there is 0% probability that the upper toxicity threshold will be exceeded. However, the upper probability bound on our Monte Carlo prediction suggests that there could be as much as 100% probability that the upper threshold will be exceeded. Therefore, selenium poses low risks to the survival and reproduction of small-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of selenium exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

TCDD-TEQs

Bayou d'Inde AOC

The Monte Carlo and lower probability bounds predictions for total daily intake rates of TCDD-TEQs by small-sized birds indicate that there is 0% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as high as 100% because of our uncertainty in model predictions.

The Monte Carlo results suggest that there is 0% probability that the upper toxicity threshold will be exceeded. However, the exceedance probability could be as high as 13% probability given the model uncertainty. Therefore, TCDD-TEQs pose low risks to the survival and reproduction of average-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of TCDD-TEQs exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and

100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo and lower probability bounds predictions for total daily intake rates of TCDD-TEQs by average-sized birds indicate that there is 0% chance that the lower toxicity threshold will be exceeded. The Monte Carlo results also suggest that there is 0% probability that the upper toxicity threshold will be exceeded. However, the upper and lower toxicity thresholds can be exceeded in some extreme cases (p=100 and 18%, respectively). Therefore, TCDD-TEQs pose low risks to the survival and reproduction of small-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of TCDD-TEQs exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 100% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Middle Calcasieu River AOC

Residue data were missing for this Area of Concern and risk characterization was not possible.

Upper Calcasieu River AOC

The Monte Carlo and lower probability bounds predictions for total daily intake rates of TCDD-TEQs by small-sized birds indicate that there is 0% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as high as 100% because of our uncertainty in model predictions (Table H1-4).

The Monte Carlo analysis results suggests that there is no chance that the upper toxicity threshold will be exceeded. However, according to the upper probability bound, there is 5% chance that the upper threshold will be exceeded. Therefore, TCDD-TEQs pose low risks to the survival and reproduction of average-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of TCDD-TEQs exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 63% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

The Monte Carlo and lower probability bounds predictions for total daily intake rates of TCDD-TEQs by average-sized birds indicate that there is 0% chance that the lower toxicity threshold will be exceeded. That probability, however, could be as high as 100% because of our uncertainty in model predictions (Table H1-4).

The Monte Carlo analyses results suggest that there is no chance that the upper toxicity threshold will be exceeded. However, the upper probability bound indicates that the exposure could exceed the upper threshold in 18% of the cases. Therefore, TCDD-TEQs pose low risks to the survival and reproduction of small-sized sediment-probing birds, but there is some uncertainty about the risk conclusion (Table H1-4).

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of TCDD-TEQs exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure have a 0% and 74% probability of exceeding the Appendix G benchmark, respectively (Table H1-5).

Reference Areas

The Monte Carlo and lower probability bounds analysis results for total daily intake rates of TCDD-TEQs by small-sized birds indicate that there is no chance that the lower toxicity threshold will be exceeded. However, under extreme exposure conditions as indicated by the upper probability bound, there could be 12% probability of exceeding the lower threshold. Despite this, small-sized birds feeding and reproducing in BI AOC should be free from adverse effects due to TCDD-TEQ exposure (Table H1-4). Therefore, TCDD-TEQs pose low risks to the survival and reproduction of average-sized sediment-probing birds (Table H1-4).

The Monte Carlo analysis indicates that average-sized sediment-probing birds have 0% probability of total daily intake of TCDD-TEQs exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure also have a 0% probability of exceeding the Appendix G benchmark (Table H1-5).

The Monte Carlo and lower probability bounds analysis results for total daily intake rates of TCDD-TEQs by average-sized birds indicate that there is no chance that the lower toxicity threshold will be exceeded. However, the upper probability bound suggests that the exceedance probability for the lower threshold could be as high as 26%. Therefore, TCDD-TEQs pose low risks to the survival and reproduction of small-sized sediment-probing birds.

The Monte Carlo analysis indicates that small-sized sediment-probing birds have 0% probability of total daily intake of TCDD-TEQs exceeding the Appendix G benchmark. The lower and upper probability bounds of exposure also have a 0% probability of exceeding the Appendix G benchmark (Table H1-5).

Review of historical data associated with CH2M Hill's Calcasieu Estuary Biological Monitoring Program indicated no invertebrate data for COC's that screened through for this assessment. Most of the reported data were for fish, which are not a part of the diet of sediment-probing birds.

4.0 Uncertainty Analysis

There are a number of sources of uncertainty in assessments of risk to sediment-probing birds, including uncertainties in the conceptual model, in the exposure assessment, and in the effects assessment. As each of these sources of uncertainty can influence the estimations of risk, it is important to describe and, when possible, quantify the magnitude and direction of such uncertainties. In this way, it is possible to evaluate the level of confidence that can be placed in the assessments conducted using the various lines of evidence.

Uncertainties Associated with the Conceptual Model - The conceptual model is intended to define the linkages between stressors, potential exposure, and predicted effects on ecological receptors. As such, the conceptual model provides the scientific basis for selecting assessment and measurement endpoints to support the risk assessment process. Potential uncertainties arise from lack of knowledge regarding ecosystem functions, failure to adequately address spatial and temporal variability in the evaluations of sources, fate, and effects, omission of stressors, and overlooking secondary effects (USEPA 1998a). The types of uncertainties associated with the conceptual model that links contaminant sources to effects on sediment-probing birds include those associated with the identification of COCs, environmental fate and transport of COCs, exposure pathways, receptors at risk, and ecological effects. Of these, the identification of exposure pathways probably represents the primary source of uncertainty in the conceptual model. In this assessment, it was assumed that exposure to contaminated food and sediments represents the most important pathways for exposing sediment-probing birds to COCs.

Uncertainties Associated with the Exposure Assessment - The exposure assessment is intended to describe the actual or potential co-occurrence of stressors with receptors. As such, the exposure assessment identifies the exposure pathways and the intensity and extent of contact with stressors for each receptor or group of receptors at risk. There are a number of potential sources of uncertainty in the exposure assessment, including measurement errors, extrapolation errors, and data gaps.

In this assessment, two types of measurements were used to evaluate exposure of sediment-probing birds to COCs, including chemical analyses of environmental media (i.e., whole sediment) and tissue residues in invertebrates. Analytical errors and descriptive errors represent potential sources of uncertainty.

Three approaches were used to address concerns relative to these sources of uncertainty.

First, analytical errors were evaluated using information on the accuracy, precision, and detection limits (DL) that are generated to support the Phase I and Phase II sampling programs. The results of this analysis indicated that most of the data used in this assessment met the project data quality objectives. Second, all data entry, data translation, and data manipulations were audited to ensure their accuracy. Data auditing involved 10% number-for-number checks against the primary data source initially, increasing to 100% number-for-number checks if significant errors were detected in the initial auditing step. Finally, statistical analyses of data were conducted to evaluate data distributions, identify the appropriate summary statistics to generate, and evaluate the variability in the observations. As such, measurement errors in the sediment chemistry data are considered to be of minor importantance and are unlikely to influence the results of the risk assessment.

Uncertainties in the Effects Assessment - The effects assessment is intended to describe the effects caused by stressors, link them to the assessment endpoints, and evaluate how effects change with levels (i.e., concentrations) of the various stressors. There are several sources of uncertainty in the assessment of effects including measurement errors, extrapolation errors, and data gaps.

Uncertainty in the exposure and effects assessments is also increased by data gaps. To the extent possible, this source of uncertainty was mitigated by collecting detailed information on the effects of COCs in the Calcasieu Estuary. In addition, the use of multiple lines of evidence provides a basis for minimizing the influence of data gaps on the effects assessment. Nevertheless, limitations on certain types of data makes it difficult to fully evaluate the effects of COC exposures on sediment-probing birds. Also, there was an absence of dose-response data or any other type of effects for sediment-probing birds. Such absence of effects data is a major source of uncertainty in the risk assessment for sediment-probing birds.

5.0 Conclusions

Mercury

The risk characterization results indicate that there is little chance of mercury exposure exceeding the effects thresholds for sediment-probing birds in the Calcasieu

Estuary. There is some uncertainty about this conclusion, however, for BI AOC. There is no risk associated with the reference areas. Thus, sediment-probing birds are at a relatively low risk for adverse effects associated with exposure to mercury in the Calcasieu Estuary.

Lead

There is a 100% probability of lead exposure exceeding the lower effects threshold for sediment-probing birds foraging in all areas of concern, as well as the reference areas. However, the upper effects threshold is not likely to be exceeded at any location. Thus, lead poses indeterminate risks to the survival and reproduction of sediment-probing birds.

Selenium

Similar to lead, levels of selenium are high throughout the Calcasieu Estuary, including the reference areas. Thus, there are relatively moderate probabilities of exceeding the lower bound toxicity. The probabilities of exceeding the lower threshold (as indicated by Monte Carlo analysis) range from 6% for the reference areas (average-sized birds) to 49% in the MCR AOC (small-sized birds). The probability of exceeding the upper bound toxicity threshold is zero in all locations. Therefore, selenium poses low risk in BI AOC (average-sized and small-sized birds), UCR AOC (average-sized birds), and reference areas (average-sized birds). Selenium posed indeterminate risk in MCR AOC (average-sized and small-sized birds), UCR AOC (small-sized birds), and reference areas (small-sized birds). Note, however, that there was some uncertainty about the risk conclusions.

TCDD-TEQs

Risk characterization indicates that there is no chance of TCDD-TEQs exposure exceeding the lower or upper bound toxicity thresholds for sediment-probing birds foraging in Calcasieu Estuary. Therefore, TCDD-TEQs pose low risks to the survival and reproduction of sediment-probing birds in the Calcasieu Estuary.

Probabilistic Risk Assessment Limitations

There are several limitations of the probabilistic risk analyses that influence our confidence regarding the above risk statements. These include:

- The sensitivity analyses for the Monte Carlo simulations indicated that the most important input variable was free metabolic rate (*FMR*). The *FMR* used in the analyses was based on the allometric equation from Nagy (1987). No corresponding measurements of this variable are available for sediment-probing bird species. The potential magnitude and direction of the uncertainty associated with lack of empirical data on metabolic rate are unknown. We did, however, investigate the possible consequences of the uncertainty in this variable due to model error (i.e., the error associated with the lack of fit of the allometric model that relates *FMR* to body weight). This source of uncertainty did not strongly impact our conclusions regarding risk.

- Our analysis focussed on fairly small sediment-probing birds that forage exclusively in each of the AOCs and the reference areas. Many sediment-probing bird species, however, are larger and forage over broader areas (e.g., ibis, spoonbills). We would expect risks to these bird species to be lower than for the hypothetical receptors considered in our analyses.

- The effects analyses pointed out several key sources of uncertainty. First, no data were available for any sediment-probing bird species. Second, differing environmental conditions between the laboratory and the field introduces uncertainty to the estimation of effects doses.

The above described limitations are common to wildlife risk assessments and indicate the value of having other lines of evidence to help characterize risks. Biological surveys and ambient toxicity testing are two such lines of evidence. No *in situ* or whole media feeding studies were available, however, for sediment-probing birds in the Calcasieu estuary. Formal biological surveys that relate degree of COC contamination to abundances of different sediment-probing bird species have not been conducted. However, bird banding and other surveys indicate that many species of sediment-probing birds (e.g., sandpipers, stilts, ibis, spoonbills and others) are common throughout the estuary. While this evidence certainly cannot be used to rule out the possibility that COCs are causing adverse effects to sediment-probing birds in locally contaminated areas, it does seem unlikely that COCs are causing widespread impacts to sediment-probing birds on a larger spatial scale.

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Tables

Table H1-1. Risk quotients for contaminants of concern.

Contaminant of Concern (COC)	Area	Risk Quotient	Proceed to Probabilistic Assessment?
Mercury	Bayou d'Inde	9	Yes
	Middle Calcasieu River	10	Yes
	Upper Calcasieu River	11	Yes
	Reference Areas	5	Yes
Lead	Bayou d'Inde	5.9	Yes
	Middle Calcasieu River	2.3	Yes
	Upper Calcasieu River	4	Yes
	Reference Areas	1.7	Yes
Selenium	Bayou d'Inde	2.3	Yes
	Upper Calcasieu River	2.8	Yes
	Middle Calcasieu River	3.7	Yes
	Reference Areas	2.1	Yes
TCDD – TEQs	Bayou d'Inde	8.9	Yes
	Upper Calcasieu River	1.9	Yes
	Middle Calcasieu River	0.892	No
	Reference Areas	0.063	Yes

Table H1-2. Monte Carlo analysis input variables.

Variable			Distribution	Parameters
Body Weight: average-sized species (*BW*; kg)			normal	Mean = 0.126, SD = 0.018
Body Weight: small-sized species (*BW*; kg)			normal	Mean = 0.044, SD = 0.0066
Free Metabolic Rate: average-sized and small species (*FMR*; Kcal/kg bw/day)			$FMR = aBW^b$	
a = FMR-slope			normal	Mean = 0.681, SD = 0.102
b = FMR-power			normal	Mean = 0.749, SD = 0.037
Gross Energy (GE_i; Kcal/kg)			lognormal	Mean = 1066, SD = 224
Assimilation Efficiency (AE_i; Unitless)			beta	alpha = 20, beta = 6.5, scale = 1.0
Contaminant of Concern (COC) - Input for Monte Carlo				
COCs	**Area**	**Tissue Classification**		
Mercury (mg/kg ww)	Bayou d'Inde	C_{inv}	lognormal	Mean = 0.0037, SD = 0.0007
		C_{sed}		Mean = 0.00315, SD = 0.00018
	Middle Calcasieu River	C_{inv}	lognormal	Mean = 0.0341, SD = 0.0010
		C_{sed}		Mean = 0.00106, SD = 0.00031
	Upper Calcasieu River	C_{inv}	lognormal	Mean = 0.0221, SD = 0.0006
		C_{sed}		Mean = 0.00052, SD = 0.00003
	Reference Areas	C_{inv}	lognormal	Mean = 0.0091, SD = 0.0003
		C_{sed}		Mean = 0.00055, SD = 0.00002
Lead (mg/kg ww)	Bayou d'Inde	C_{inv}	lognormal	Mean = 0.191, SD = 0.005
		C_{sed}		Mean = 40.8, SD = 3.18
	Middle Calcasieu River	C_{inv}	lognormal	Mean = 0.097, SD = 0.002
		C_{sed}		Mean = 20.2, SD = 1.91
	Upper Calcasieu River	C_{inv}	lognormal	Mean = 0.068, SD = 0.006
		C_{sed}		Mean = 19, SD = 1.41
		C_{inv}	lognormal	Mean = 0.384, SD = 0.017
		C_{sed}		Mean = 18.1, SD = 0.184

Table H1-2. Monte Carlo analysis input variables.

Variable			Distribution	Parameters
COCs (cont.)	**Area (cont.)**	**Tissue Classification (cont.)**		
Selenium (mg/kg ww)	Bayou d'Inde	C_{inv}	lognormal	Mean = 0.324, SD = 0.002
		C_{sed}		Mean = 0.876, SD = 0.017
	Middle Calcasieu River	C_{inv}	lognormal	Mean = 0.55, SD = 0.011
		C_{sed}		Mean = 0.62, SD = 0.019
	Upper Calcasieu River	C_{inv}	lognormal	Mean = 0.492, SD = 0.007
		C_{sed}		Mean = 0.562, SD = 0.011
	Reference Areas	C_{inv}	lognormal	Mean = 0.355, SD = 0.006
		C_{sed}		Mean = 0.715, SD = 0.002
TCDD – TEQs (ng/kg ww)	Bayou d'Inde	C_{inv}	lognormal	Mean = 22.9, SD = 0.0028
		C_{sed}		NA
	Upper Calcasieu River	C_{inv}	lognormal	Mean = 13.9, SD = 0.576
		C_{sed}		NA
	Reference Areas	C_{inv}	lognormal	Mean = 4.53, SD = 0.126
		C_{sed}		NA

SD = Standard deviation; NA = Not applicable; TCDD = tetrachlorodibenzo-*p*-dioxin; TEQs = toxic equivalents.

Table H1-3. Probability Bounds analysis input variables.

Variable	Distribution	Parameters
Probability Bounds		
Body Weight: average-sized species (*BW*; kg)	normal	Mean = 0.126, SD = 0.018
Body Weight: small-sized species (*BW*; kg)	normal	Mean = 0.044, SD = 0.0066
Free Metabolic Rate: average-sized species (*FMR*; Kcal/kg bw/day)	$FMR = aBW^b$	
a = FMR-slope	normal	Mean = 0.681, SD = 0.102
b = FMR-power	normal	Mean = 0.749, SD = 0.037
Gross Energy - Fish (GE_i; Kcal/kg)	lognormal	Mean = 1066; SD = 224
Assimilation Efficiency (AE_i, unitless)	minmaxmean	0.46, 1, 0.76

Contaminant of Concern (COC) - Input for Probability Bounds

COCs	Area	Tissue Classification	Distribution	Parameters
Mercury (mg/kg ww)	Bayou d'Inde	C_{inv}	lognormal	Mean = 1.02, SD = 0.61
		C_{sed}		Mean = 0.0071, SD = 0.002
	Middle Calcasieu River	C_{inv}	lognormal	Mean = 0.0393, SD = 0.0055
		C_{sed}		Mean = 0.0023, SD = 0.0009
	Upper Calcasieu River	C_{inv}	lognormal	Mean = 0.0346, SD = 0.0055
		C_{sed}		Mean = 0.000681, SD = 0.0001
	Reference Areas	C_{inv}	lognormal	Mean = 0.0139, SD = 0.0025
		C_{sed}		Mean = 0.000781, SD = 0.0002
Lead (mg/kg ww)	Bayou d'Inde	C_{inv}	lognormal	Mean = 0.44, SD = 0.0882
		C_{sed}		Mean = 65.8, SD = 4.36
	Middle Calcasieu River	C_{inv}	lognormal	Mean = 0.16, SD = 0.04
		C_{sed}		Mean = 32, SD = 3.03
		C_{inv}	lognormal	Mean = 0.45, SD = 0.17
		C_{sed}		Mean = 28.8, SD = 2.49

Table H1-3. Probability Bounds analysis input variables.

Variable			Distribution	Parameters
COCs (cont.)	**Area (cont.)**	**Tissue Classification (cont.)**		
Lead (mg/kg ww; cont.)	Reference Areas	C_{inv}	lognormal	Mean = 0.78, SD = 0.23
		C_{sed}		Mean = 22.9, SD = 4.16
Selenium (mg/kg ww)	Bayou d'Inde	C_{inv}	lognormal	Mean = 0.43, SD = 0.06
		C_{sed}		Mean = 1.06, SD = 0.07
	Middle Calcasieu River	C_{inv}	lognormal	Mean = 0.53, SD = 093
		C_{sed}		Mean = 0.86, SD = 0.08
	Upper Calcasieu River	C_{inv}	lognormal	Mean = 0.65, SD = 0.08
		C_{sed}		Mean = 0.7, SD = 0.06
	Reference Areas	C_{inv}	lognormal	Mean = 0.35, SD = 0.64
		C_{sed}	minmax	Min=0.65, Max=0.75
TCDD – TEQs (ng/kg ww)	Bayou d'Inde	C_{inv}	lognormal	Mean = 124, SD = 97.3
		C_{sed}		NA
	Upper Calcasieu River	C_{inv}	lognormal	Mean = 46.5, SD = 53.7
		C_{sed}		NA
	Reference Areas	C_{inv}	minmax	Min=2.14, Max=9.4
		C_{sed}		NA

SD = Standard deviation; NA = Not applicable; TCDD = tetrachlorodibenzo-*p*-dioxin; TEQs = toxic equivalents.

Table H1-4. Summary of exceedance probabilities for sediment-probing birds from Calcasieu Estuary.

Location	Probability of Exceedance (%)											
	Average-Sized Sediment-Probing Birds						Small Sediment-Probing Birds					
	LB		*FOMC*		*UB*		*LB*		*FOMC*		*UB*	
	LT	**UT**	**LT**	**UT**	**LT**	**UT**	**LT**	**UT**	**LT**	**UT**	**LT**	**UT**
Mercury												
Bayou d'Inde	0	0	0	0	100	94	0	0	0	0	100	99
Middle Calcasieu River	0	0	0	0	33	0	0	0	0	0	54	0
Upper Calcasieu River	0	0	0	0	18	0	0	0	0	0	33	0
Reference Areas	0	0	0	0	0	0	0	0	0	0	4	0
Lead												
Bayou d'Inde	81	0	100	0	100	100	93	0	100	0	100	100
Middle Calcasieu River	12	0	100	0	100	100	36	0	100	0	100	100
Upper Calcasieu River	7	0	100	0	100	100	27	0	100	0	100	100
Reference Areas	0	0	100	0	100	100	4	0	100	0	100	100
Selenium												
Bayou d'Inde	0	0	3	0	100	100	0	0	10	0	100	100
Middle Calcasieu River	0	0	23	0	100	100	0	0	49	0	100	100
Upper Calcasieu River	0	0	16	0	100	100	0	0	37	0	100	100
Reference Areas	0	0	6	0	75	12	20	0	18	2	100	67
TCDD − TEQs												
Bayou d'Inde	0	0	0	0	100	13	0	0	0	0	100	18
Upper Calcasieu River		0	0	0	100	5	0	0	0	0	100	6
Reference Areas	0	0	0	0	12	0	0	0	0	0	26	0

LB = Lower Probability Bound; FOMC = First Order Monte Carlo; UB = Upper Probability Bound; LT = Lower Toxicity Threshold; UT = Upper Toxicity Threshold; TCDD = tetrachlorodibenzo-*p* -dioxin; TEQs = toxic equivalents.

Table H1-5. Summary of exceedance probabilities for the screening level ecological risk assessment Appendix G benchmarks.

Location	Benchmark	Probability of Exceedance (%)					
		Average-Sized Sediment-Probing Birds			Small Sediment-Probing Birds		
		LB	*FOMC*	*UB*	*LB*	*FOMC*	*UB*
Mercury							
Bayou d'Inde	0.0202 mg/kg bw/d	0	0	100	0	0	100
Middle Calcasieu River		0	0	100	0	0	100
Upper Calcasieu River		0	0	100	0	0	100
Reference Areas		0	0	43	0	9	66
Lead							
Bayou d'Inde	3.53 mg/kg bw/d	0	4	100	0	14	100
Middle Calcasieu River		0	0	100	0	0	100
Upper Calcasieu River		0	0	100	0	0	100
Reference Areas		0	0	100	0	0	100
Selenium							
Bayou d'Inde	0.707 mg/kg bw/d	0	0	100	0	0	100
Middle Calcasieu River		0	0	100	0	0	100
Upper Calcasieu River		0	0	100	0	0	100
Reference Areas		0	0	100	0	0	100
TCDD -- TEQs							
Bayou d'Inde	44.3 ng/kg bw/d	0	0	100	0	0	100
Upper Calcasieu River		0	0	63	0	0	74
Reference Areas		0	0	0	0	0	0

LB = Lower Probability Bound; FOMC = First Order Monte Carlo; UB = Upper Probability Bound; LT = Lower Toxicity Threshold; UT = Upper Toxicity Threshold; TCDD = tetrachlorodibenzo-*p* -dioxin; TEQs = toxic equivalents.

Figures

Figure H1-1. Overview of approach used to assess exposure of sediment-probing birds to contaminants of concern (COCs) in the Calcasieu Estuary.

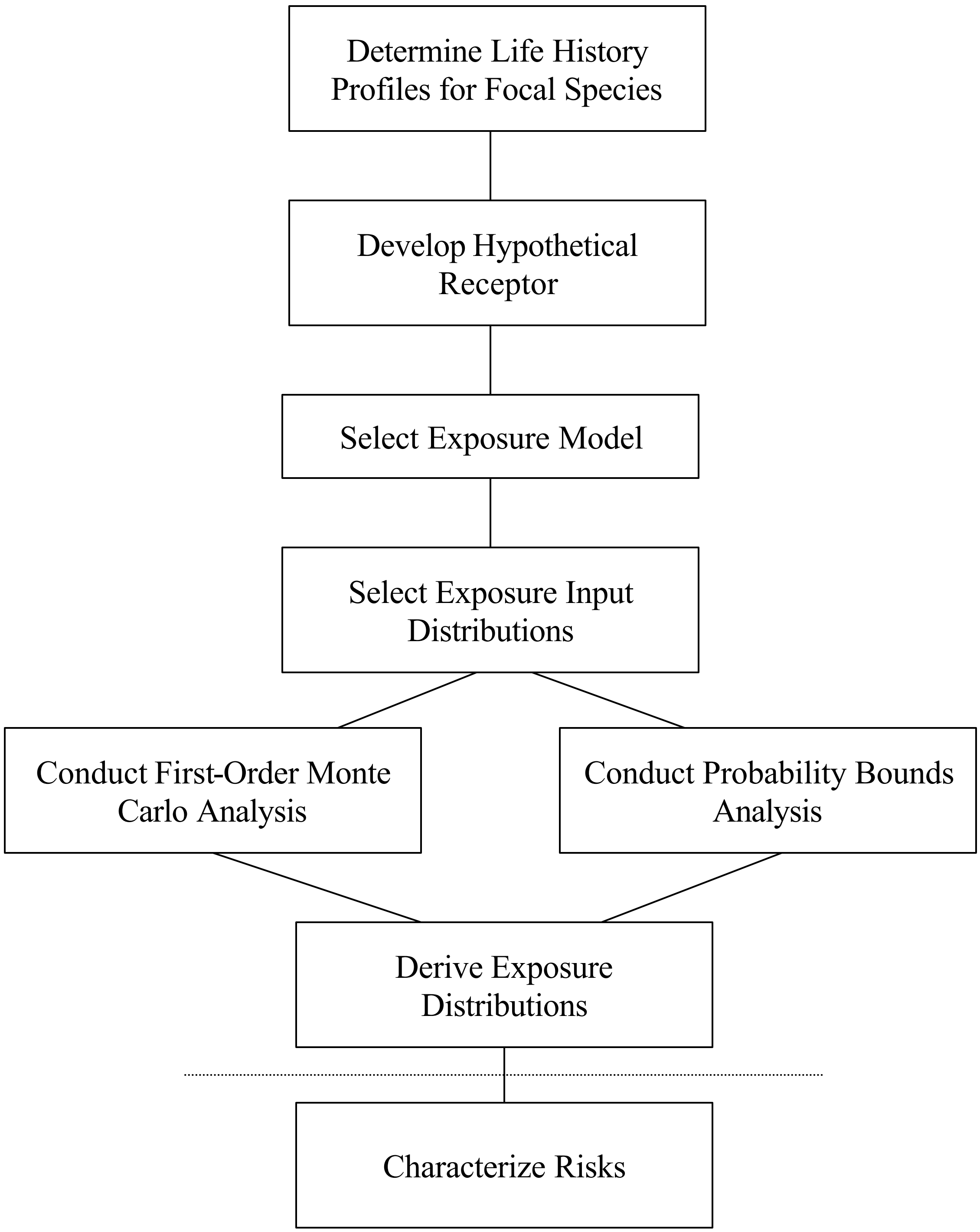

Figure H1-2. Overview of approach used to assess the effects to sediment-probing birds exposed to contaminants of concern (COCs) in the Calcasieu Estuary.

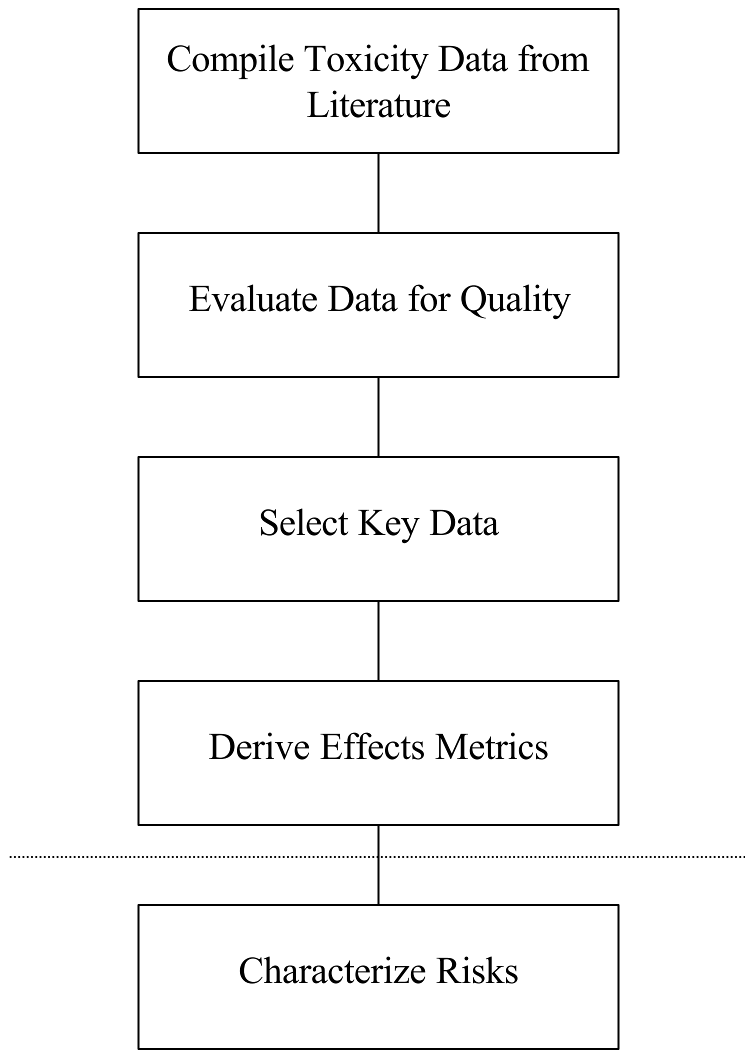

Figure H1-3. Overview of approach used to assess the risks to sediment-probing birds exposed to contaminants of concern (COCs) in the Calcasieu Estuary.

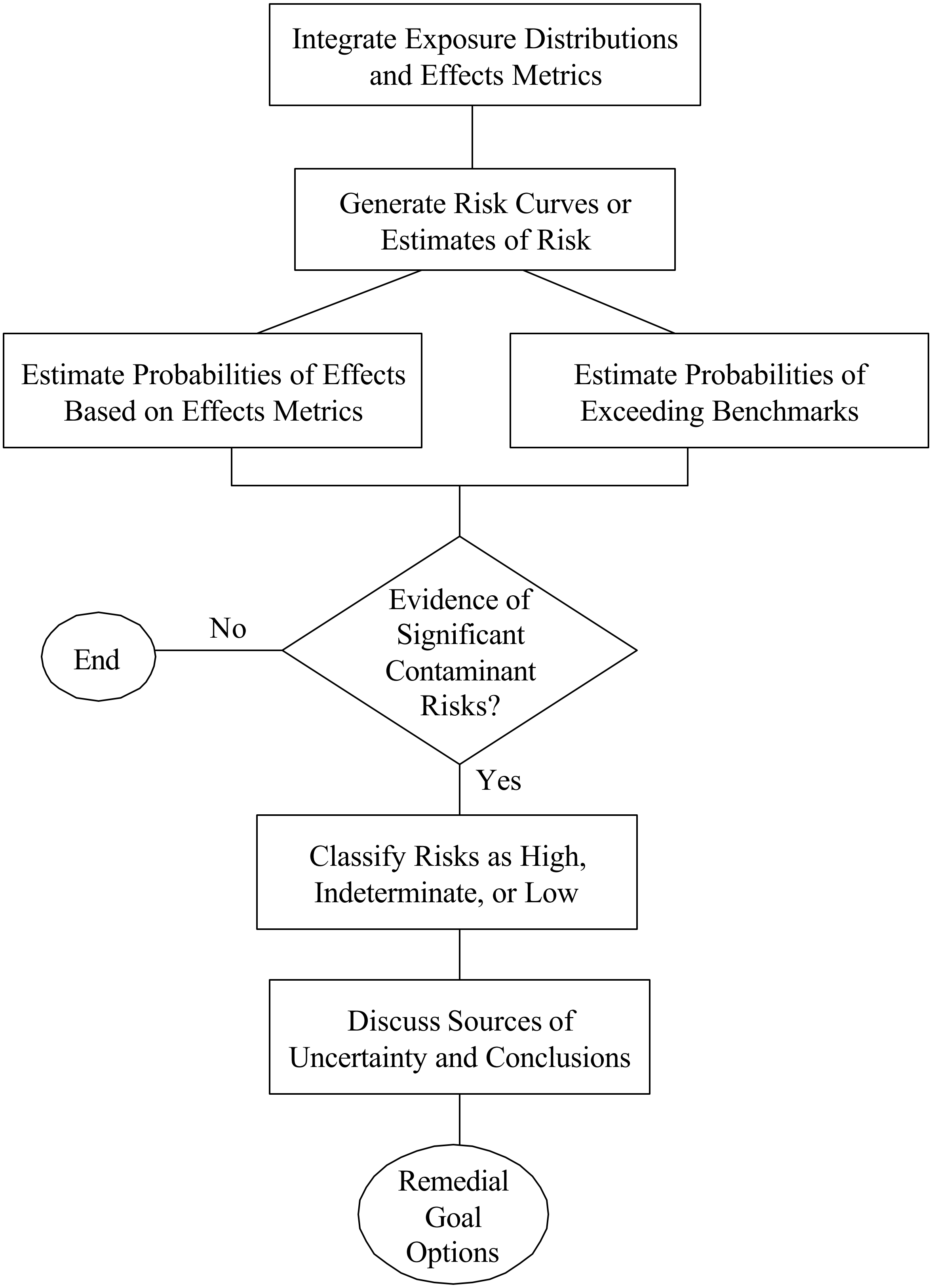

Figure H1-4. Reverse cumulative probability distribution of total daily intake rates of mercury by average-sized sediment-probing birds in Bayou d'Inde, Calcasieu Estuary.

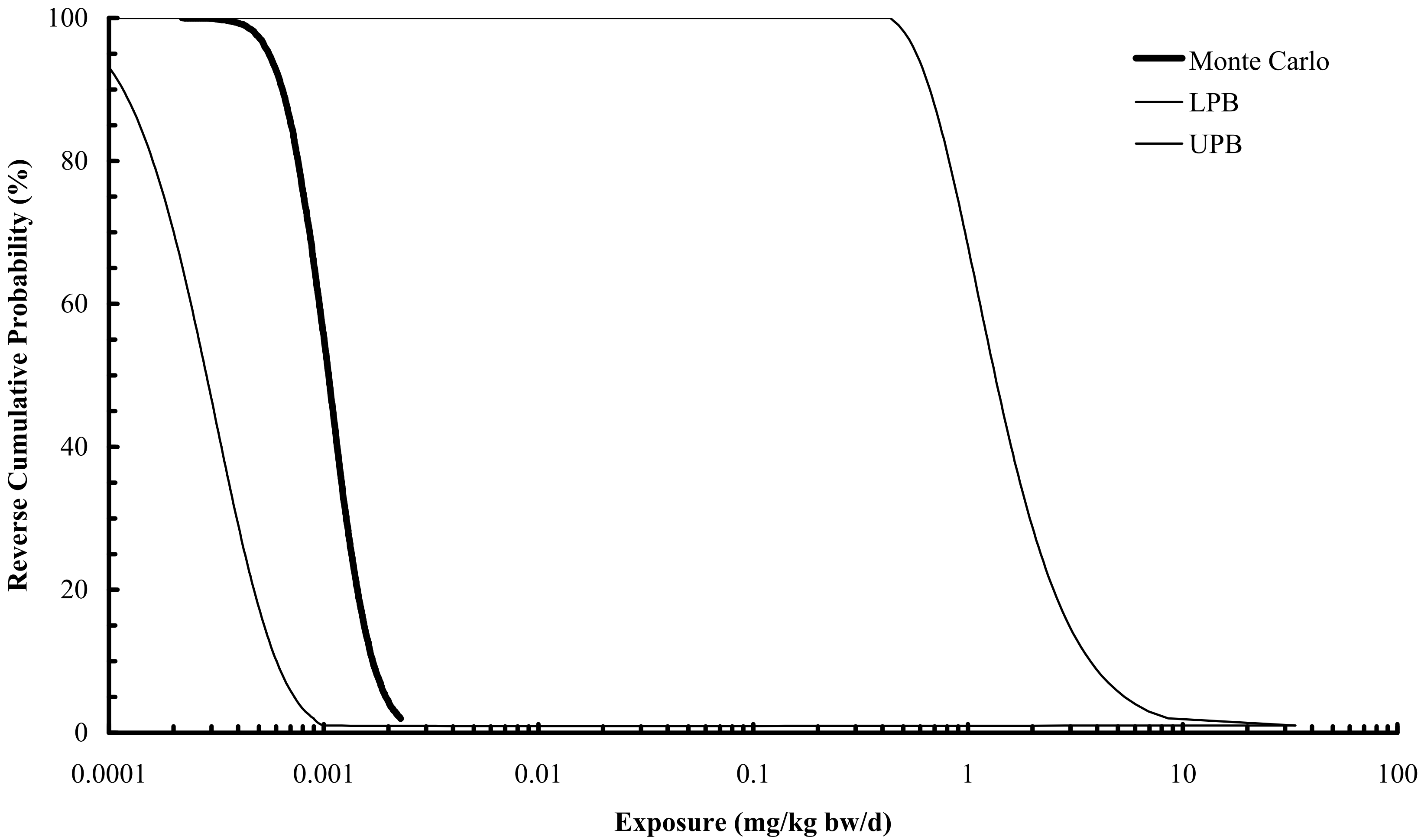

Figure H1-5. Reverse cumulative probability distribution of total daily intake rates of mercury by small sediment-probing birds in Bayou d'Inde, Calcasieu Estuary.

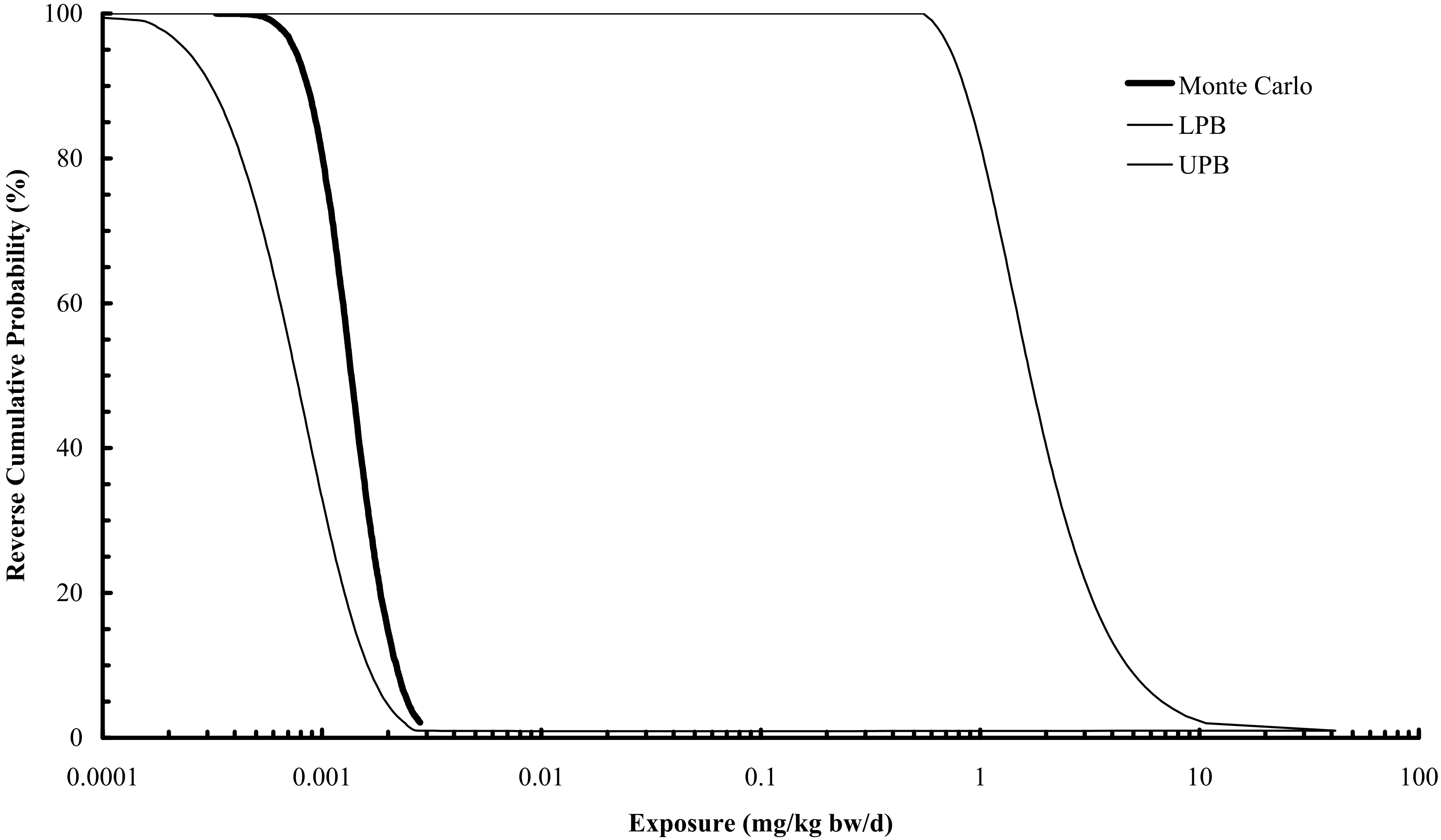

Figure H1-6. Reverse cumulative probability distribution of total daily intake rates of mercury by average-sized sediment-probing birds in the Middle Calcasieu River, Calcasieu Estuary.

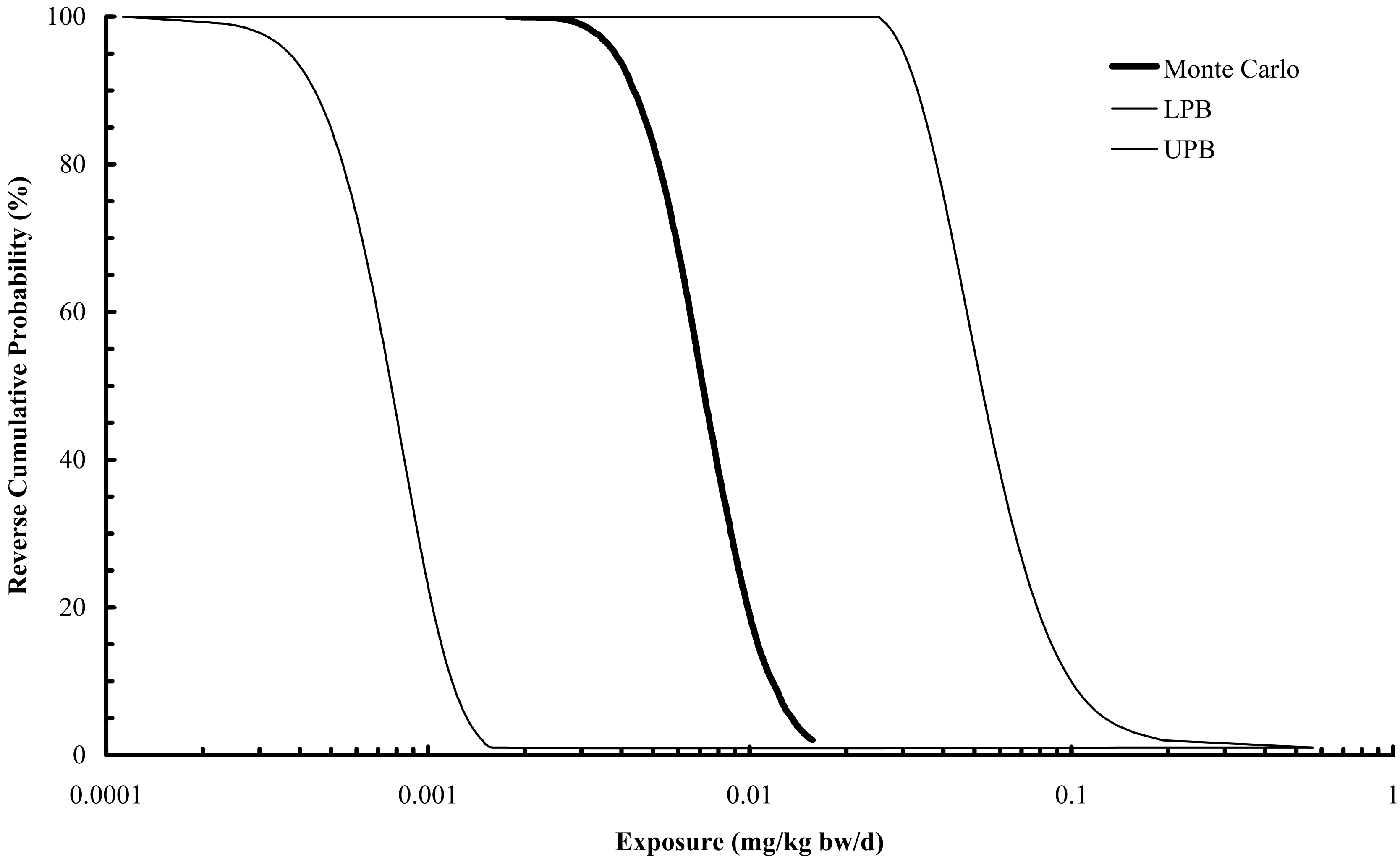

Figure H1-7. Reverse cumulative probability distribution of total daily intake rates of mercury by small sediment-probing birds in the Middle Calcasieu River, Calcasieu Estuary.

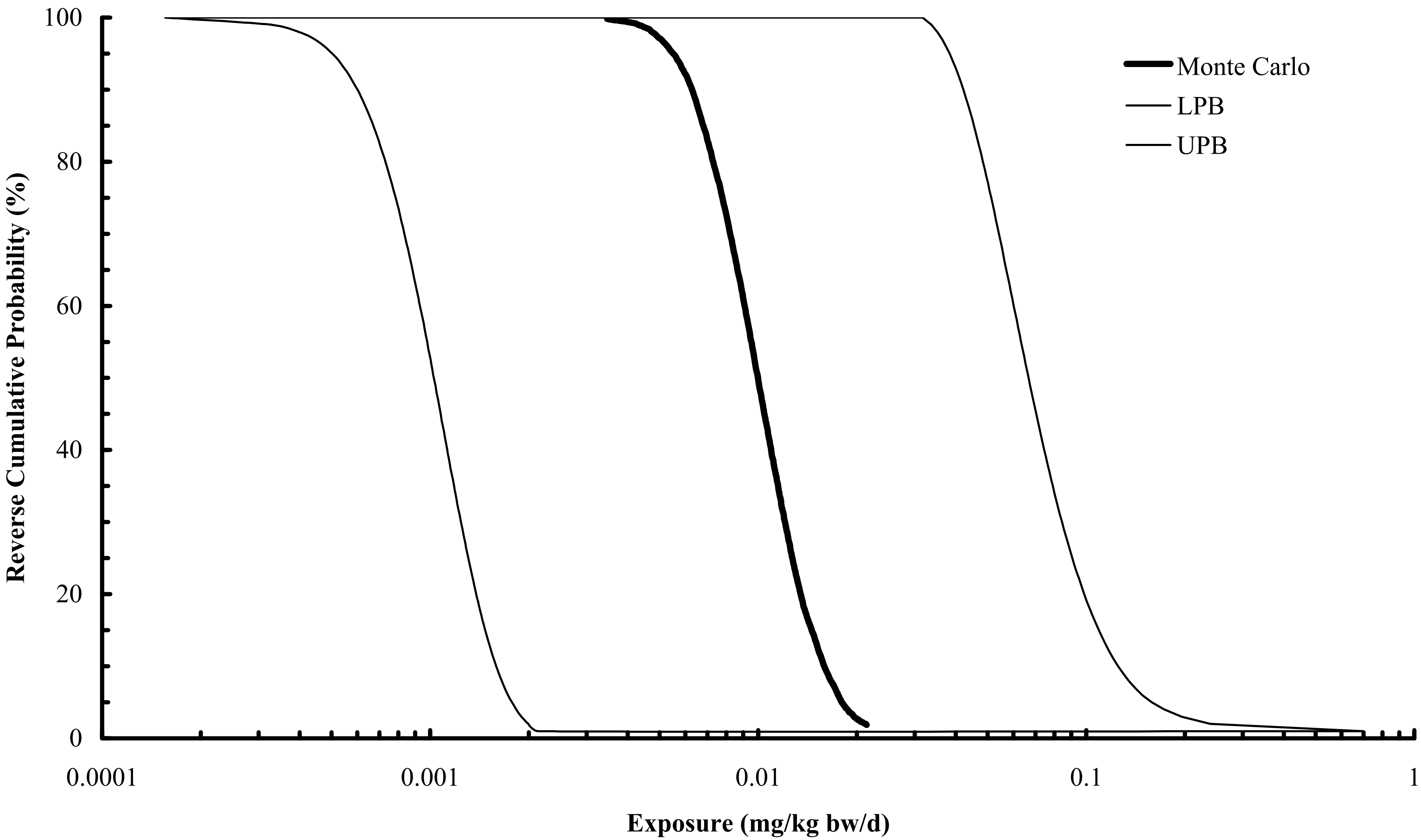

Figure H1-8. Reverse cumulative probability distribution of total daily intake rates of mercury by average-sized sediment-probing birds in the Upper Calcasieu River, Calcasieu Estuary.

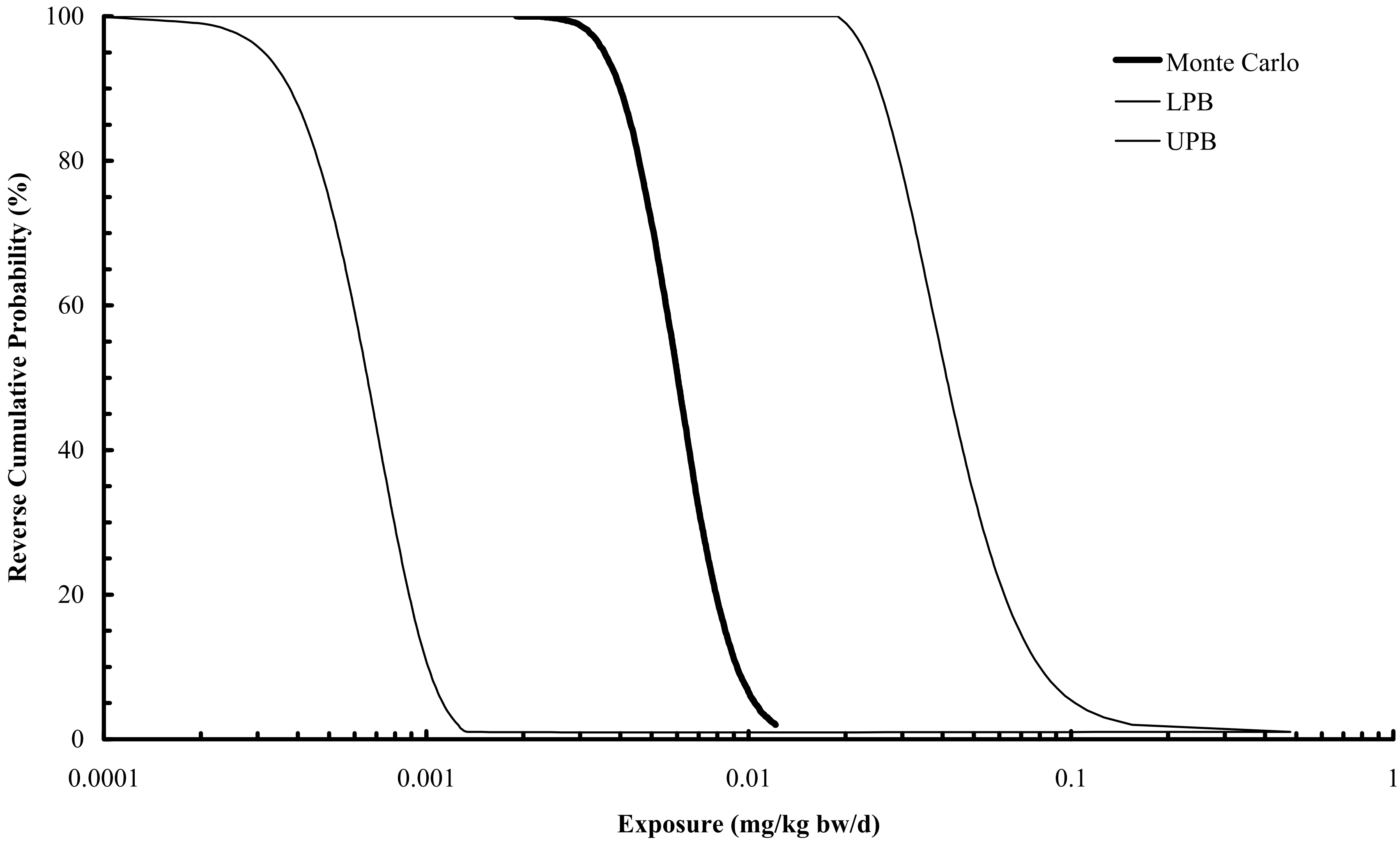

Figure H1-9. Reverse cumulative probability distribution of total daily intake rates of mercury by small sediment-probing birds in the Upper Calcasieu River, Calcasieu Estuary.

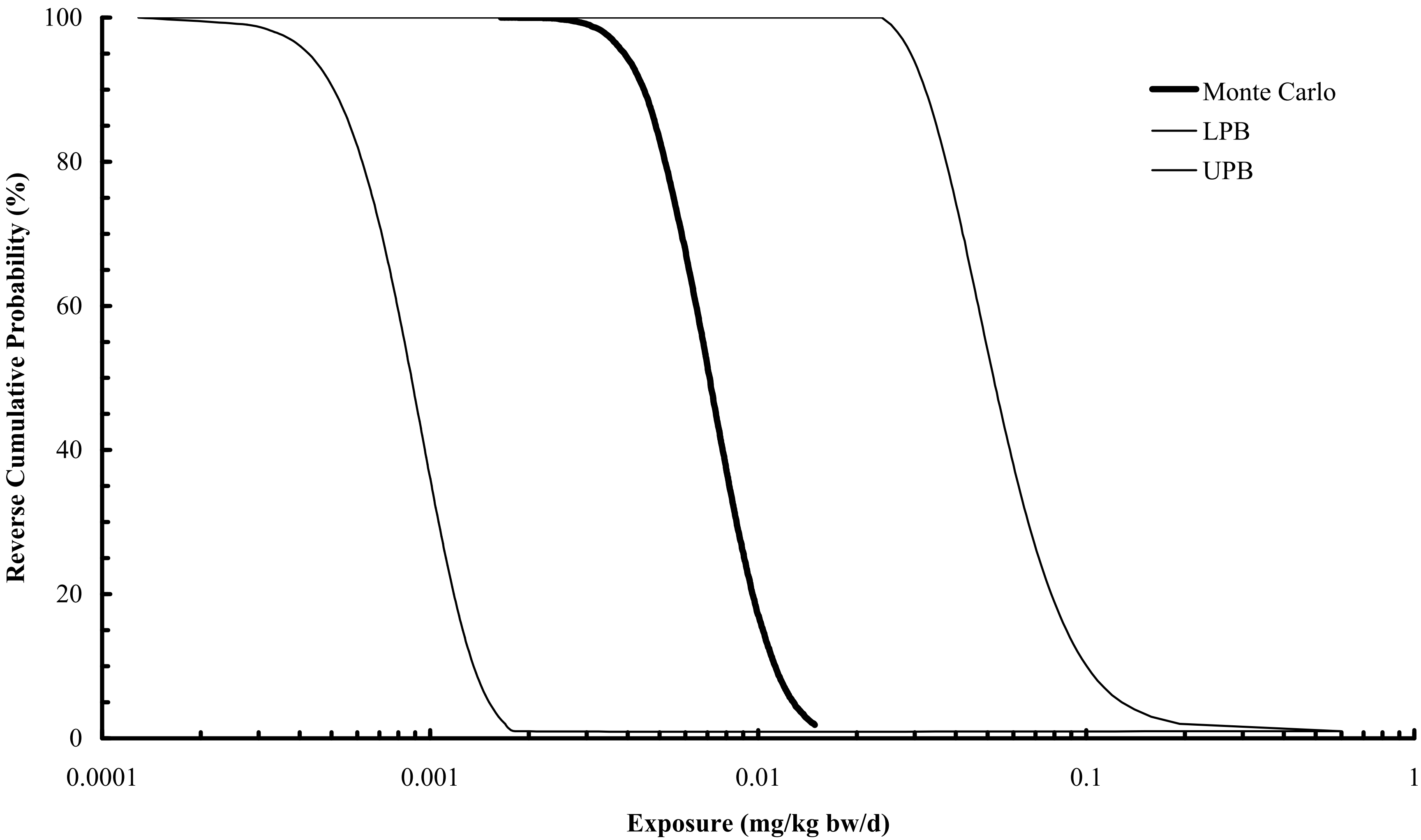

Figure H1-10. Reverse cumulative probability distribution of total daily intake rates of mercury by average-sized sediment-probing birds in the reference areas, Calcasieu Estuary.

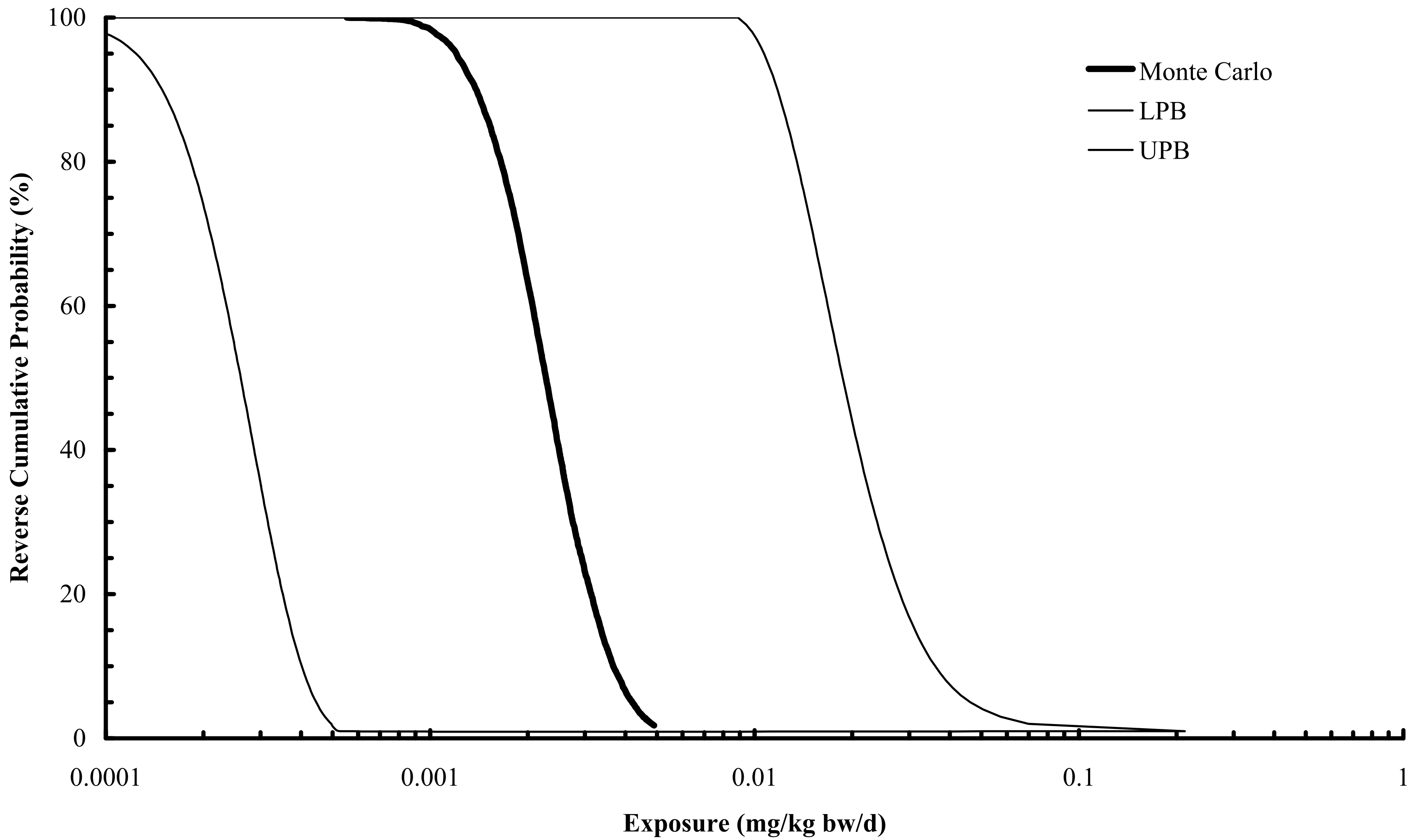

Figure H1-11. Reverse cumulative probability distribution of total daily intake rates of mercury by small sediment-probing birds in the reference areas, Calcasieu Estuary.

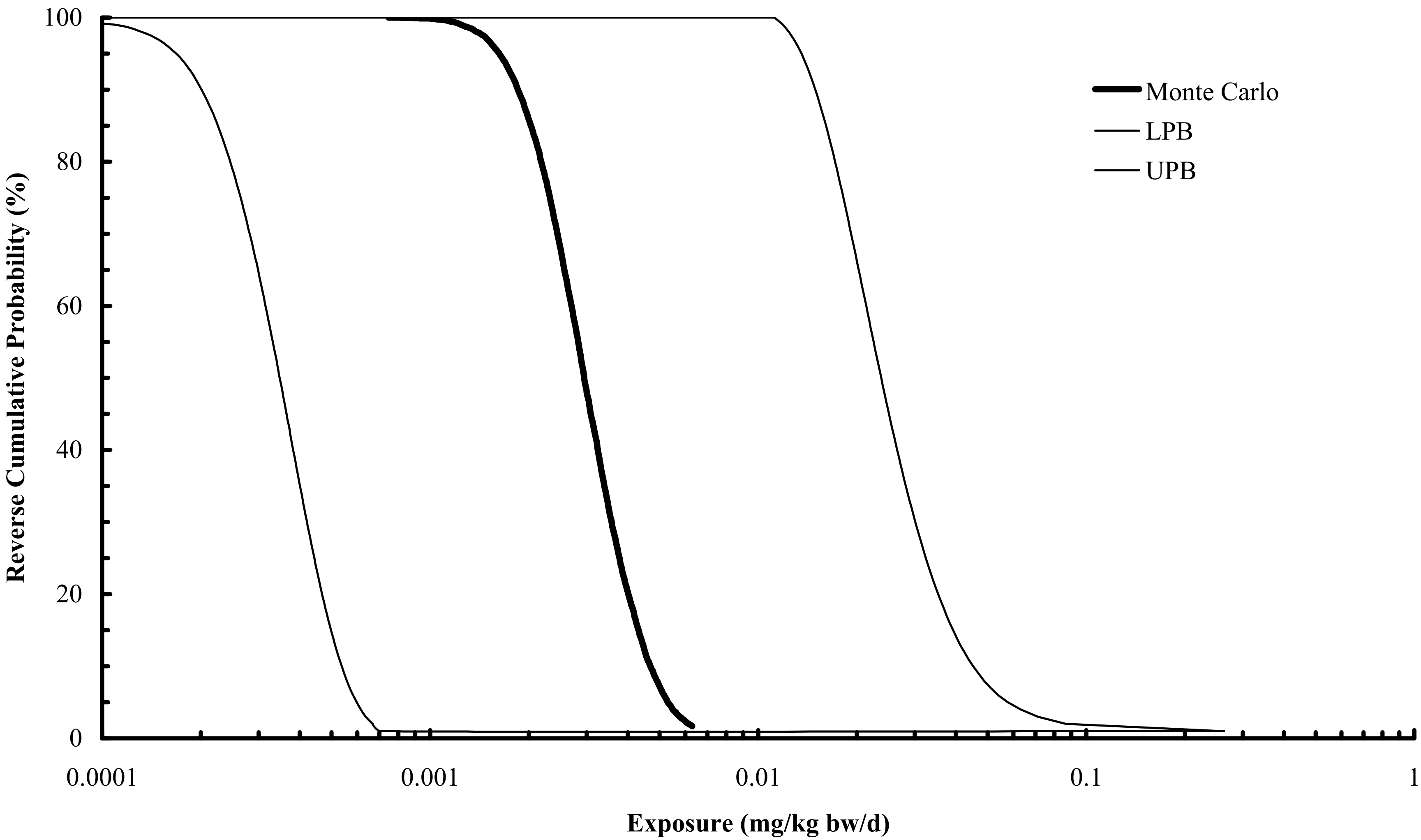

Figure H1-12. Reverse cumulative probability distribution of total daily intake rates of lead by average-sized sediment-probing birds in Bayou d'Inde, Calcasieu Estuary.

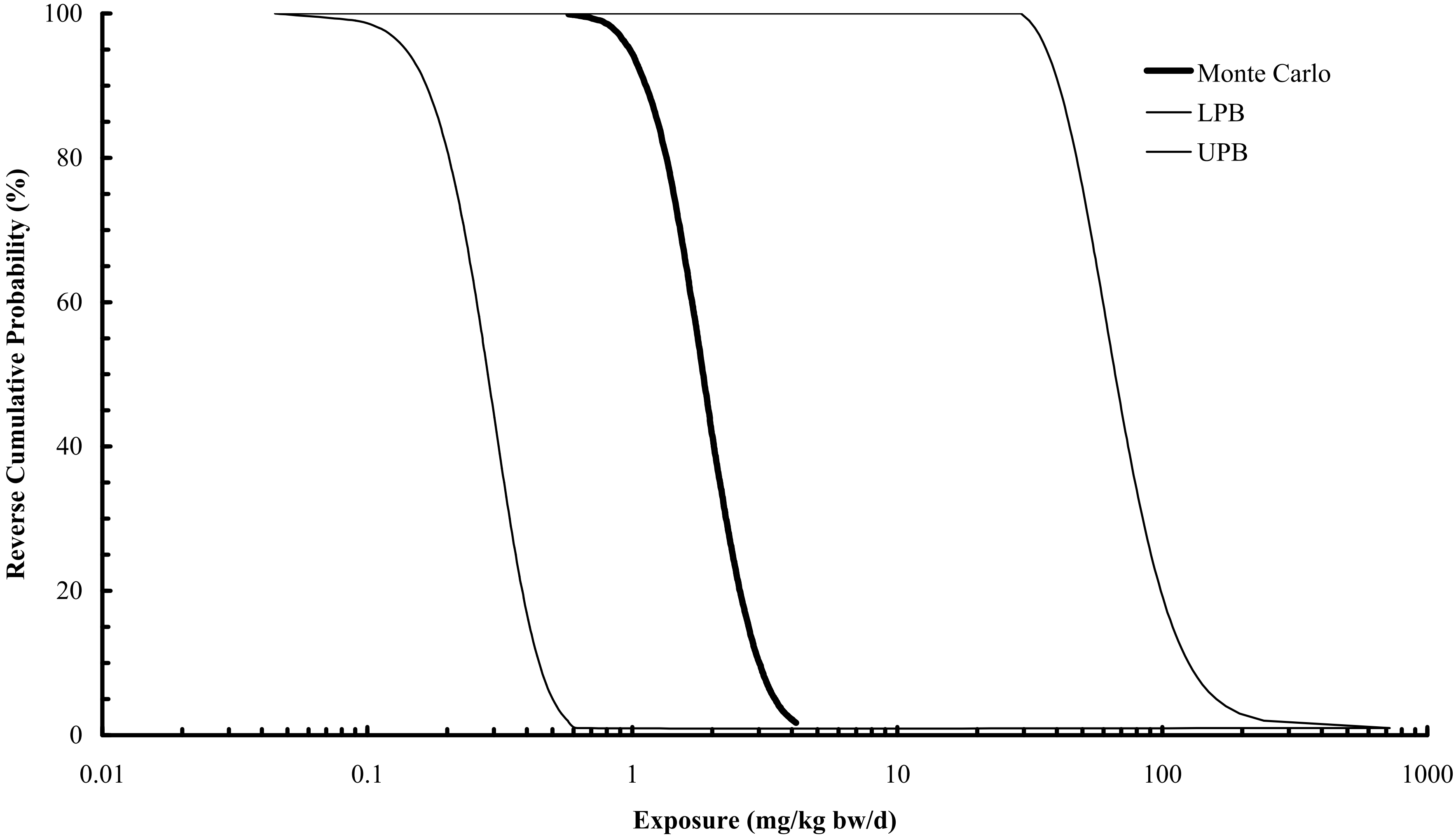

Figure H1-13. Reverse cumulative probability distribution of total daily intake rates of lead by small sediment-probing birds in Bayou d'Inde, Calcasieu Estuary.

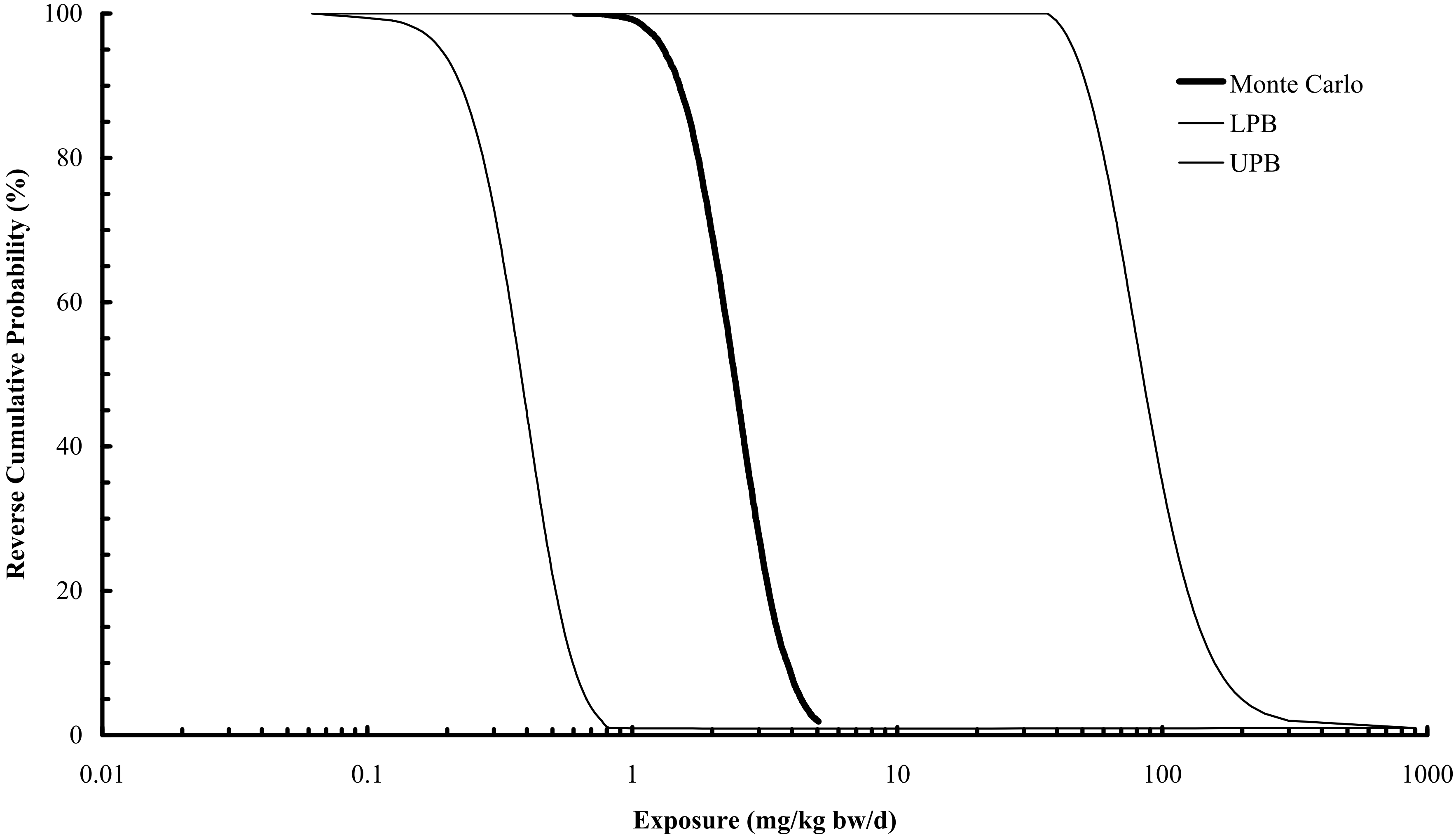

Figure H1-14. Reverse cumulative probability distribution of total daily intake rates of lead by average-sized sediment-probing birds in the Middle Calcasieu River, Calcasieu Estuary.

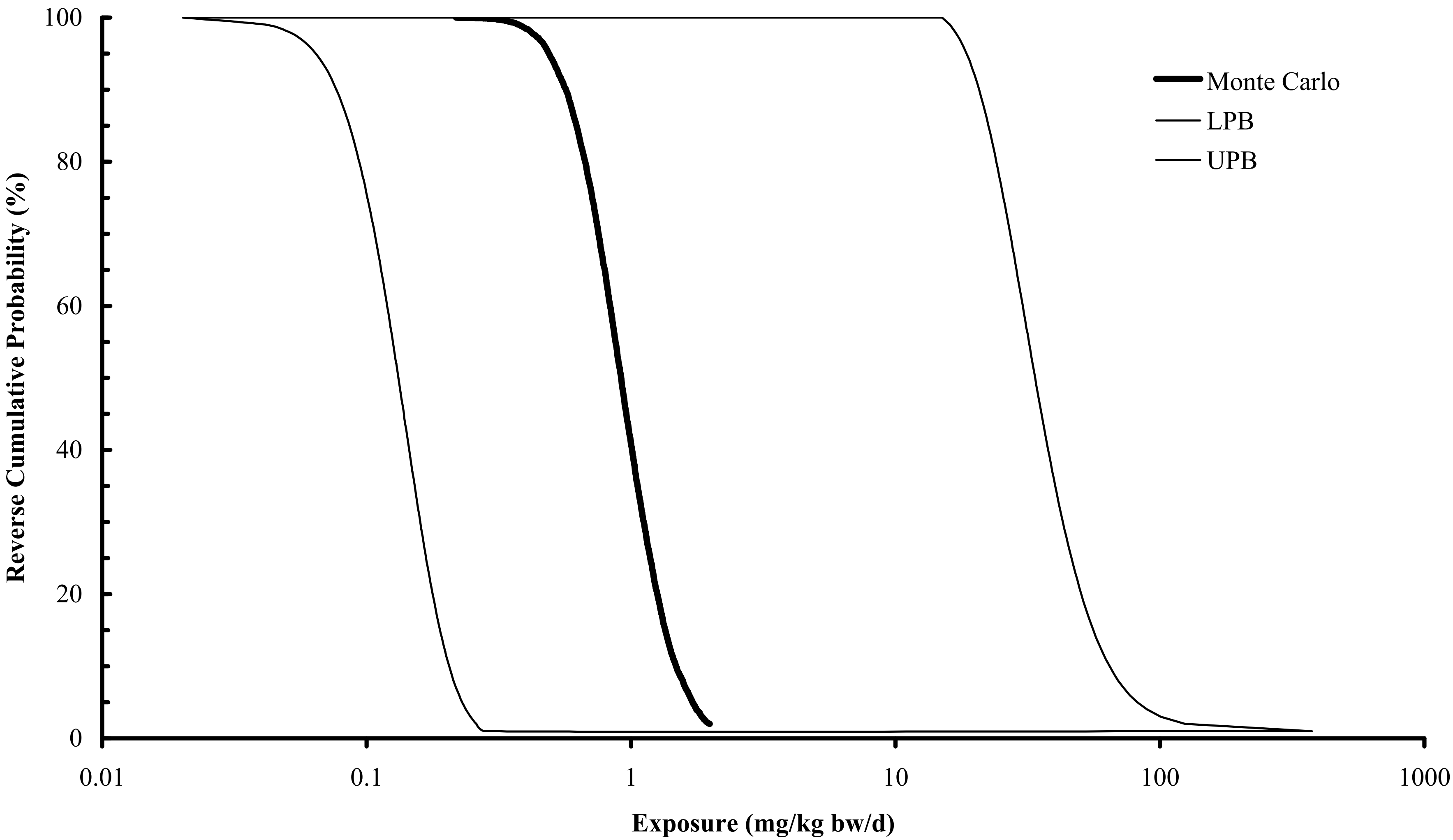

Figure H1-15. Reverse cumulative probability distribution of total daily intake rates of lead by small sediment-probing birds in the Middle Calcasieu River, Calcasieu Estuary.

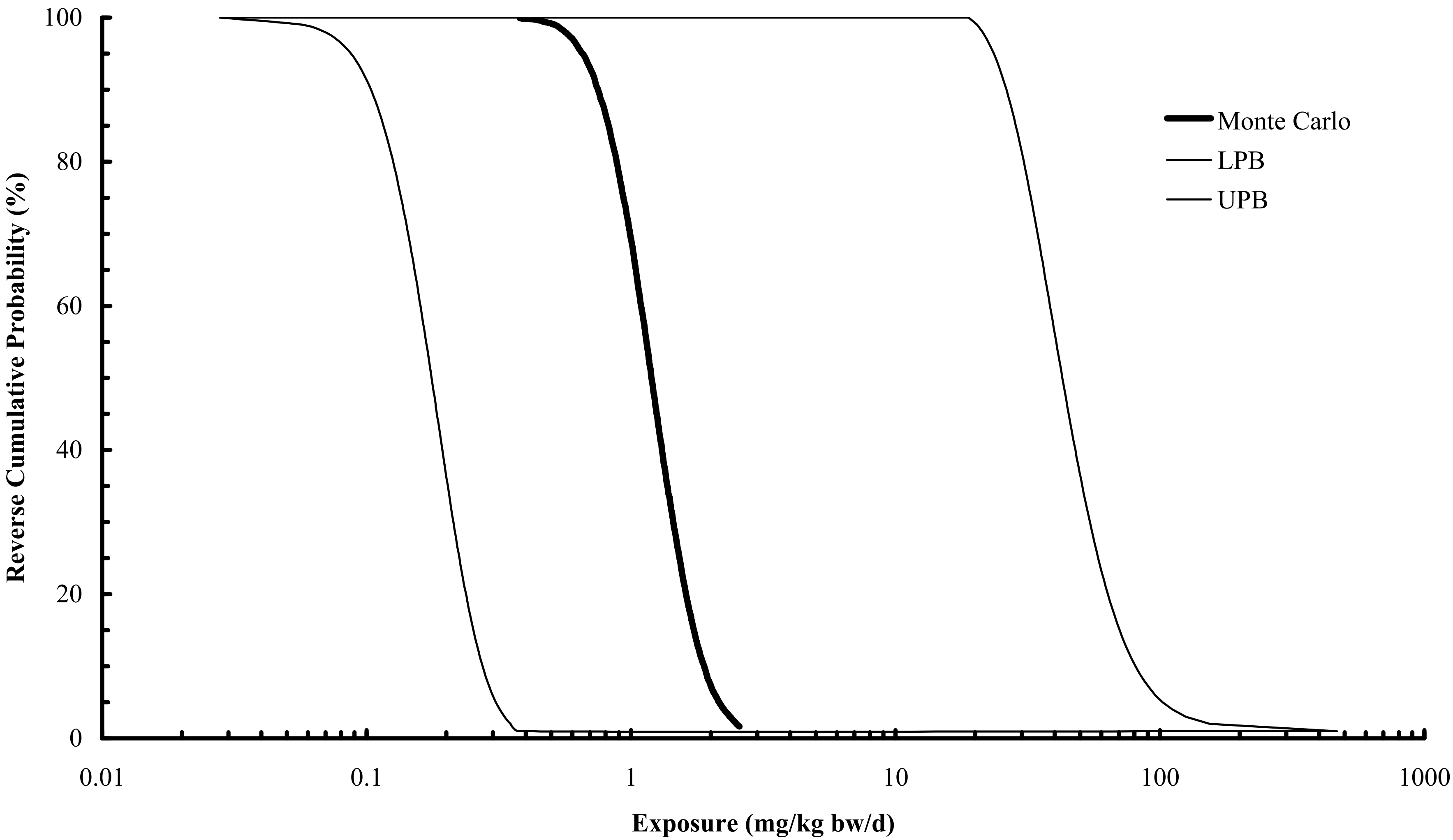

Figure H1-16. Reverse cumulative probability distribution of total daily intake rates of lead by average-sized sediment-probing birds in the Upper Calcasieu River, Calcasieu Estuary.

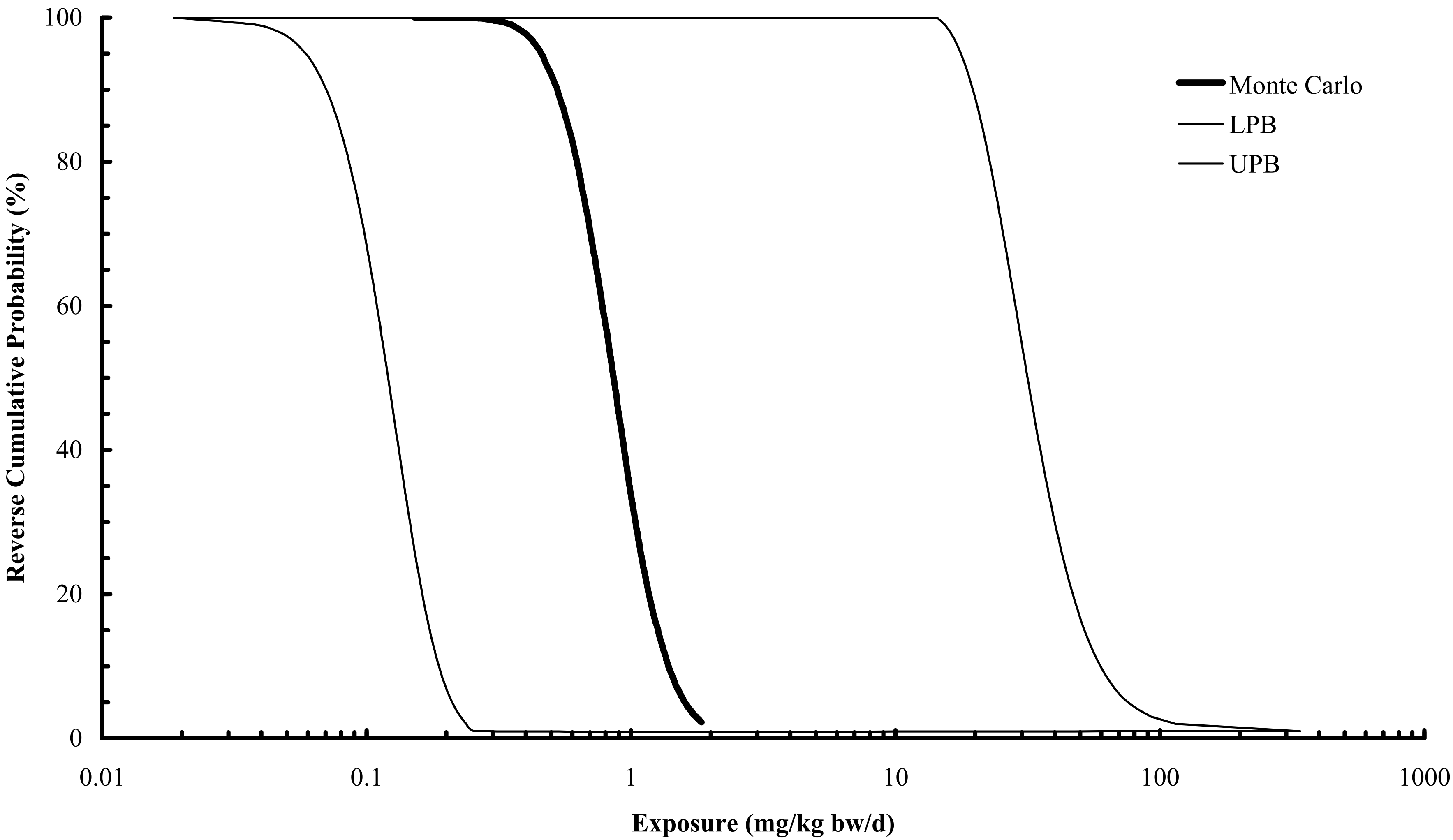

Figure H1-17. Reverse cumulative probability distribution of total daily intake rates of lead by small sediment-probing birds in the Upper Calcasieu River, Calcasieu Estuary.

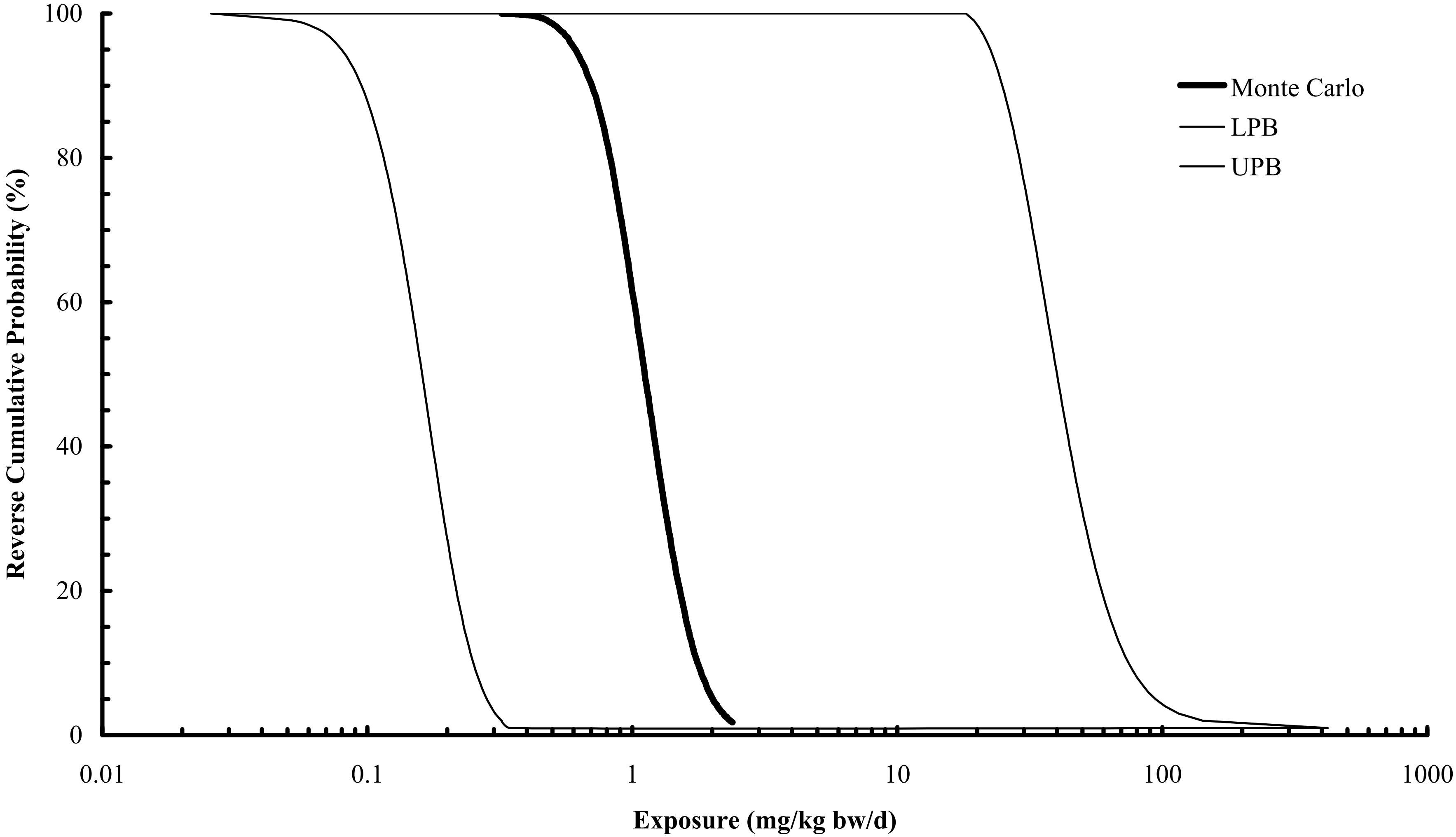

Figure H1-18. Reverse cumulative probability distribution of total daily intake rates of lead by average-sized sediment-probing birds in the reference areas, Calcasieu Estuary.

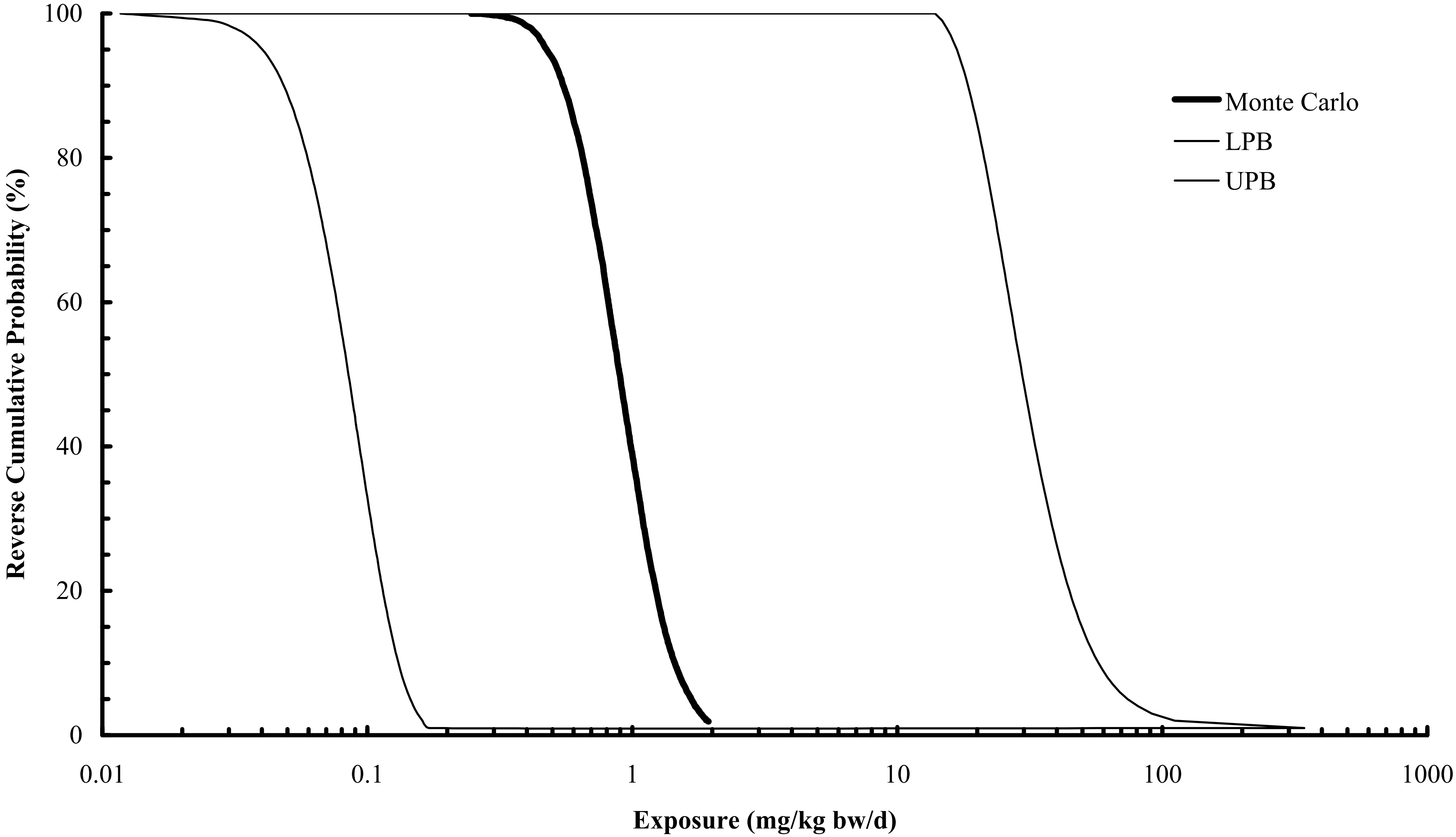

Figure H1-19. Reverse cumulative probability distribution of total daily intake rates of lead by small sediment-probing birds in the reference areas, Calcasieu Estuary.

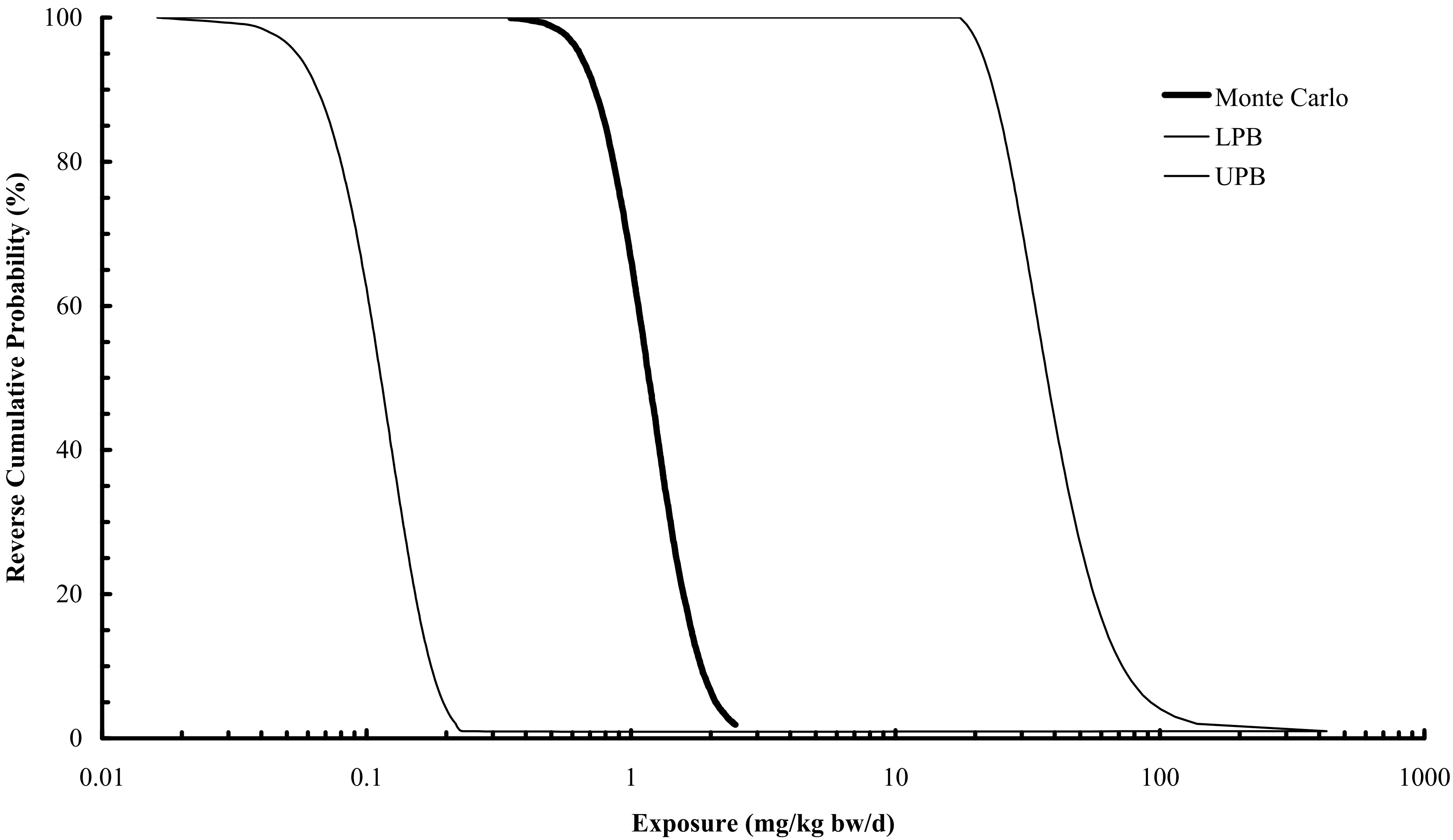

Figure H1-20. Reverse cumulative probability distribution of total daily intake rates of selenium by average-sized sediment-probing birds in Bayou d'Inde, Calcasieu Estuary.

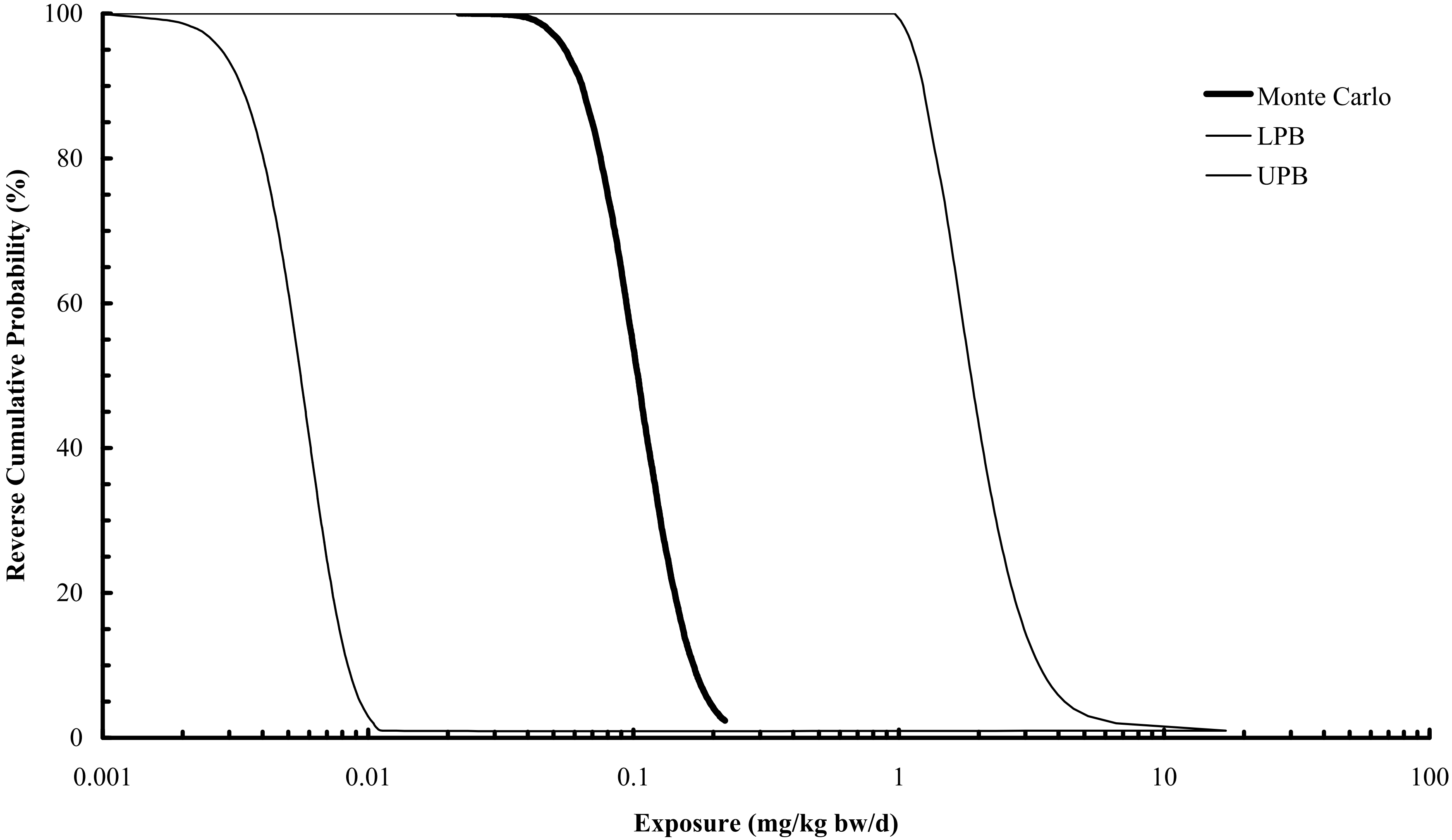

Figure H1-21. Reverse cumulative probability distribution of total daily intake rates of selenium by small sediment-probing birds in Bayou d'Inde, Calcasieu Estuary.

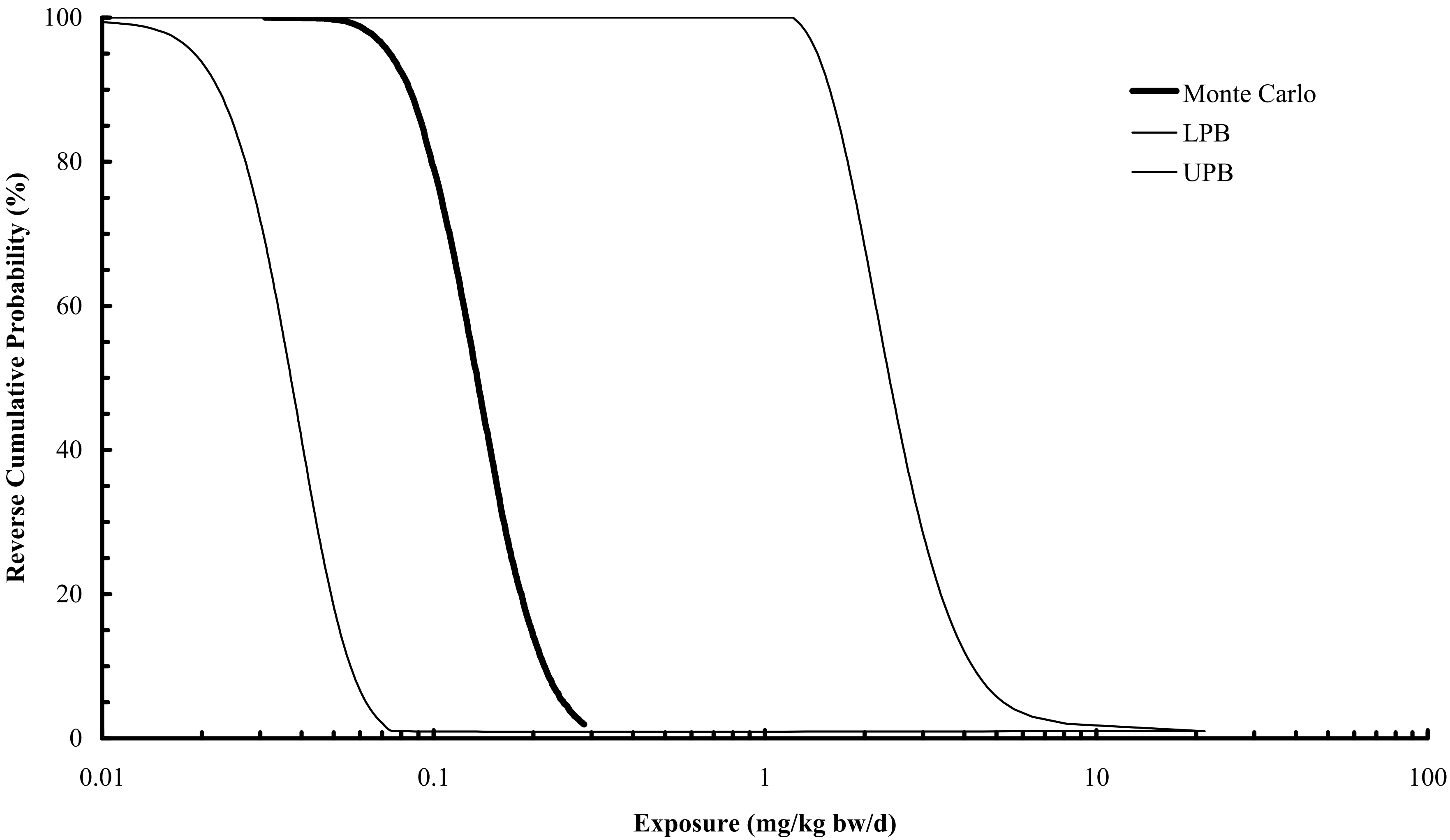

Figure H1-22. Reverse cumulative probability distribution of total daily intake rates of selenium by average-sized sediment-probing birds in the Middle Calcasieu River, Calcasieu Estuary.

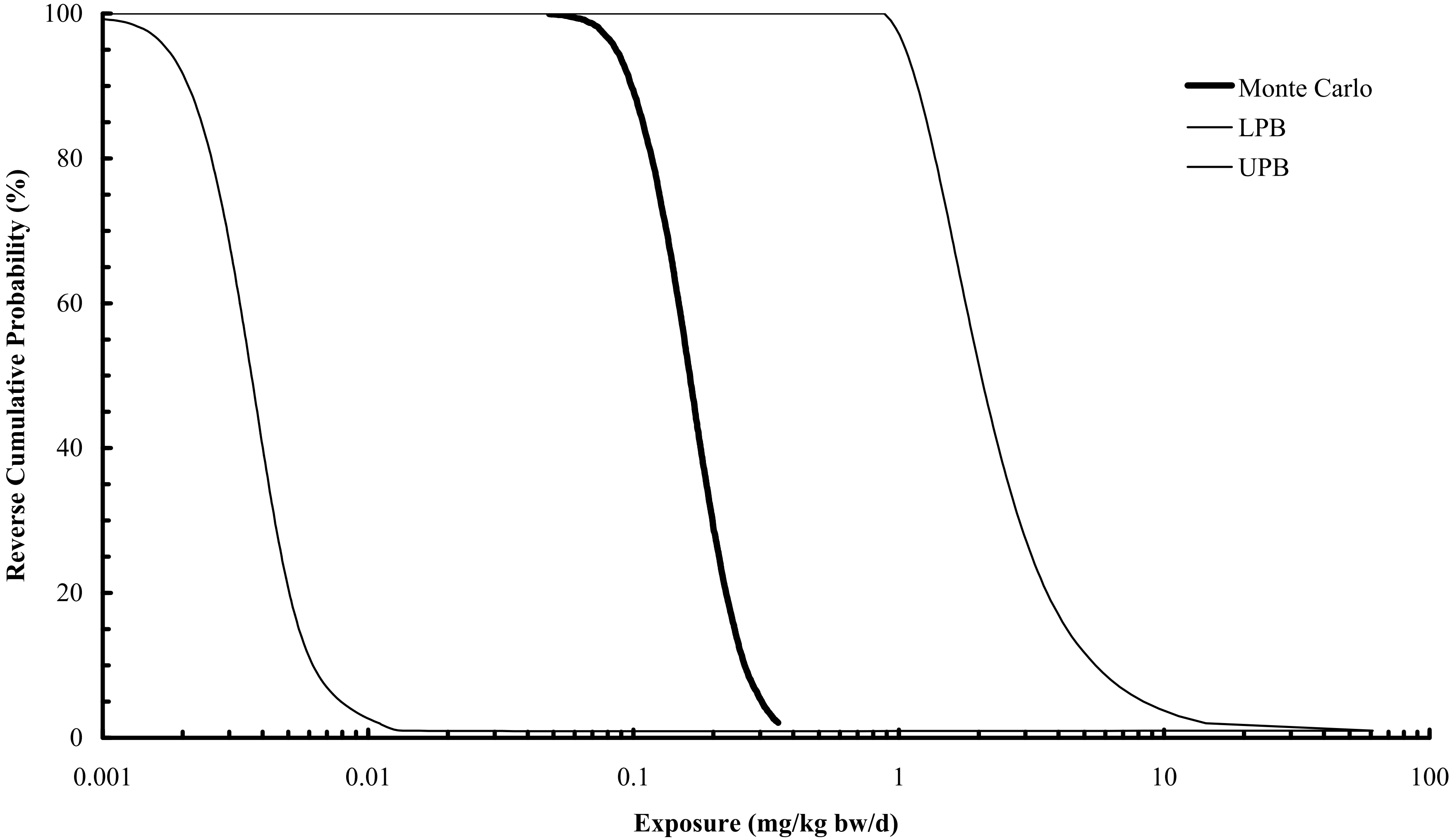

Figure H1-23. Reverse cumulative probability distribution of total daily intake rates of selenium by small sediment-probing birds in the Middle Calcasieu, Calcasieu Estuary.

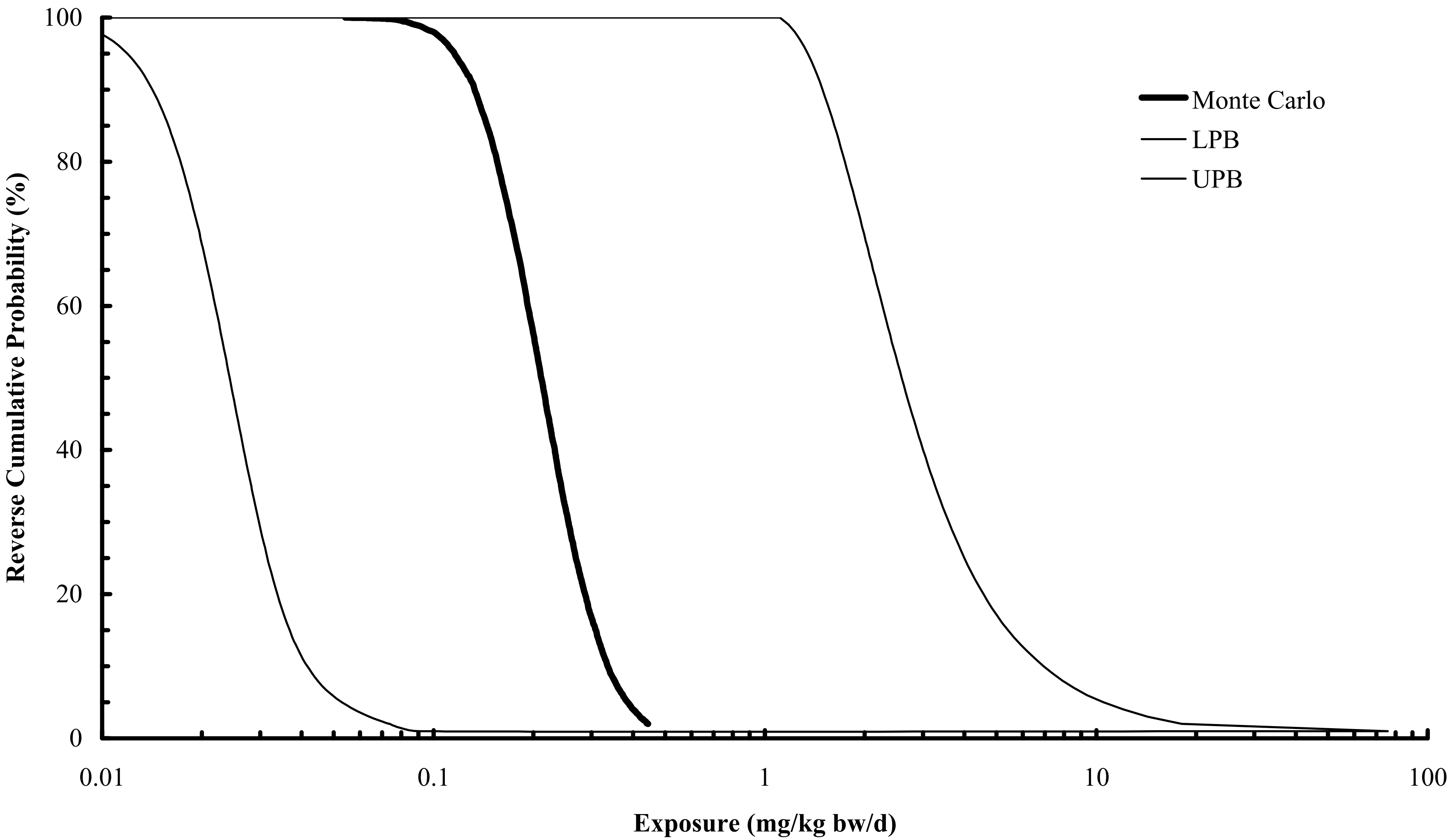

Figure H1-24. Reverse cumulative probability distribution of total daily intake rates of selenium by average-sized sediment-probing birds in the Upper Calcasieu River, Calcasieu Estuary.

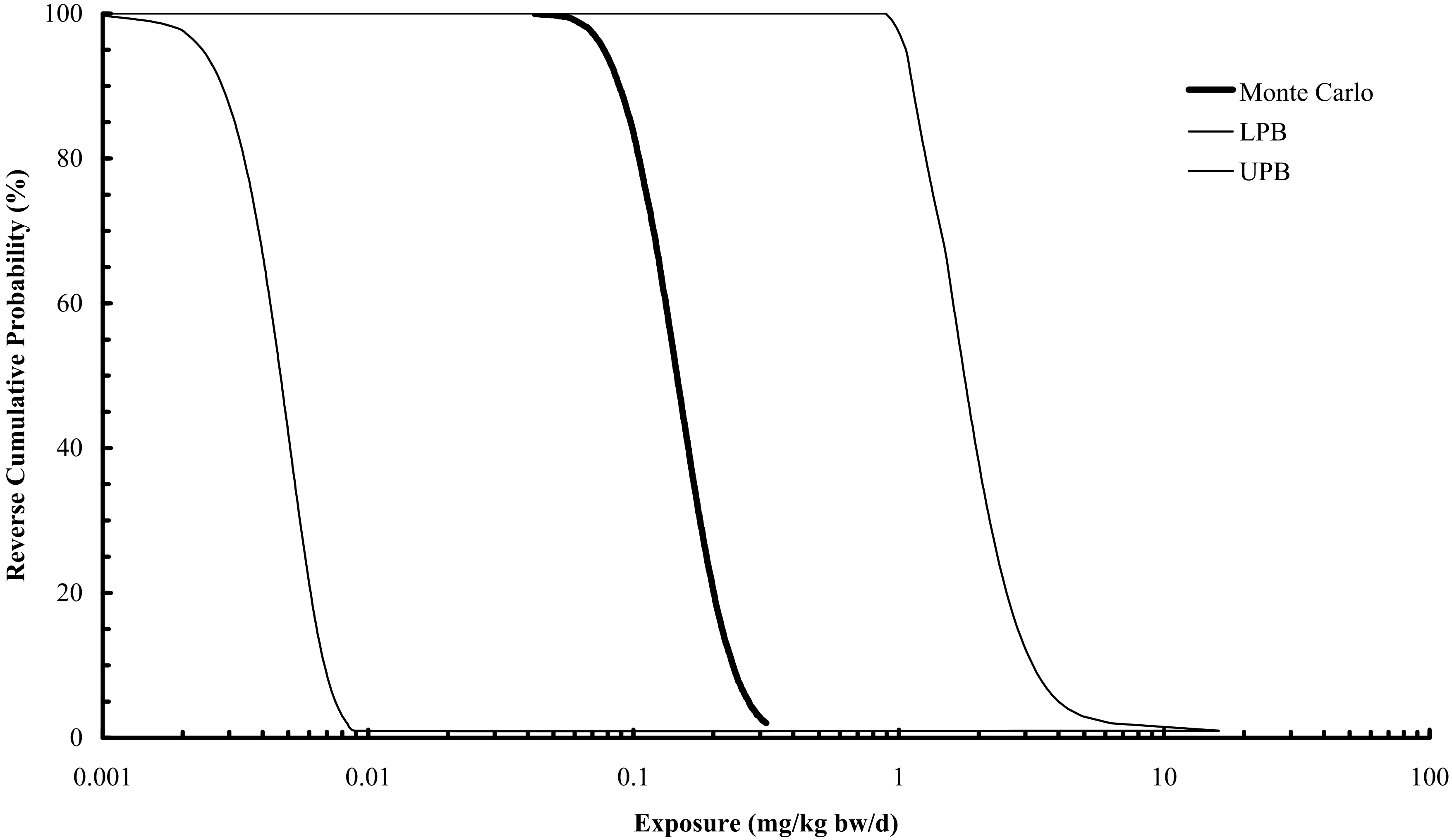

Figure H1-25. Reverse cumulative probability distribution of total daily intake rates of selenium by small sediment-probing birds in the Upper Calcasieu River, Calcasieu Estuary.

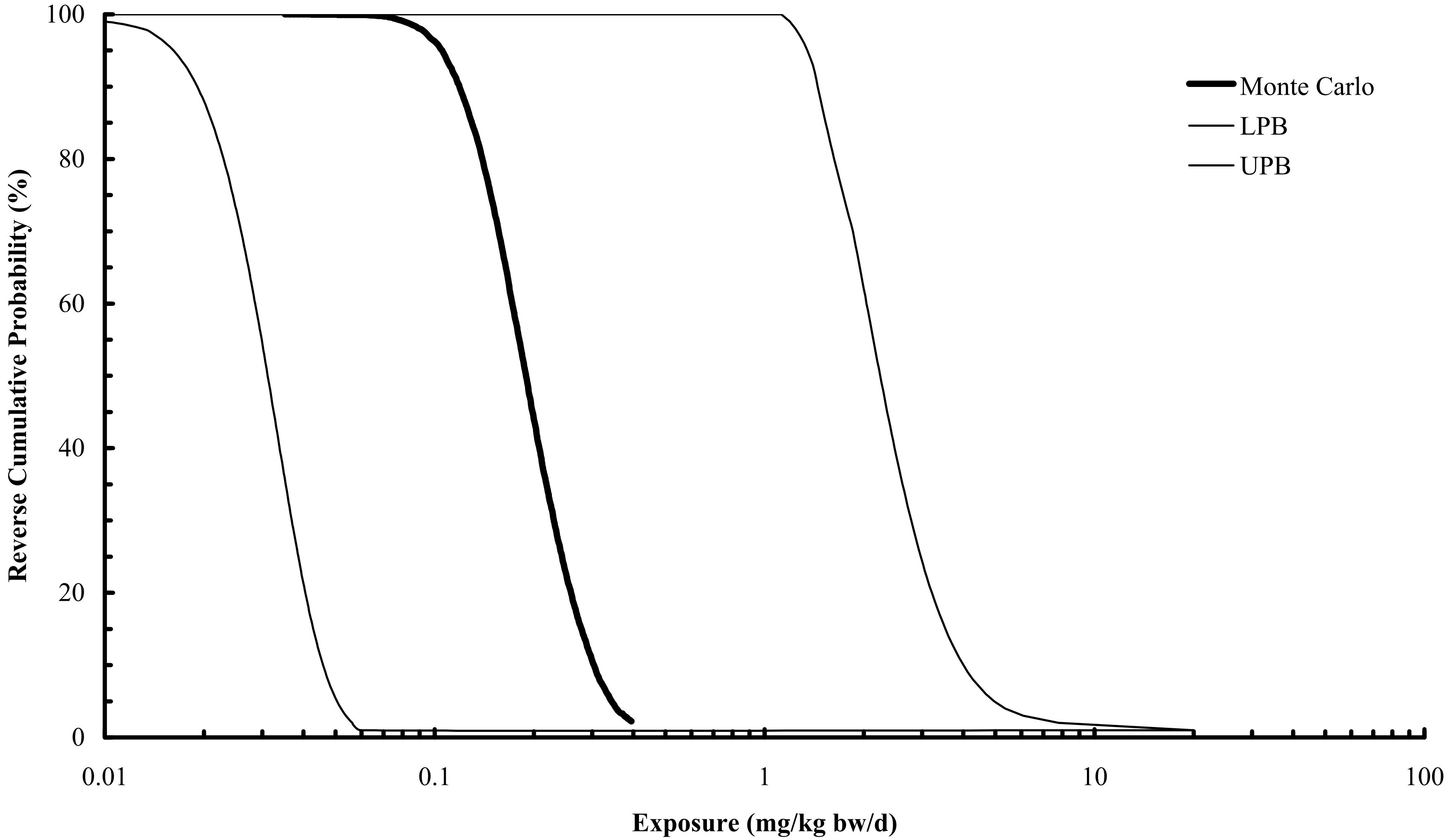

Figure H1-26. Reverse cumulative probability distribution of total daily intake rates of selenium by average-sized sediment-probing birds in the reference areas, Calcasieu Estuary.

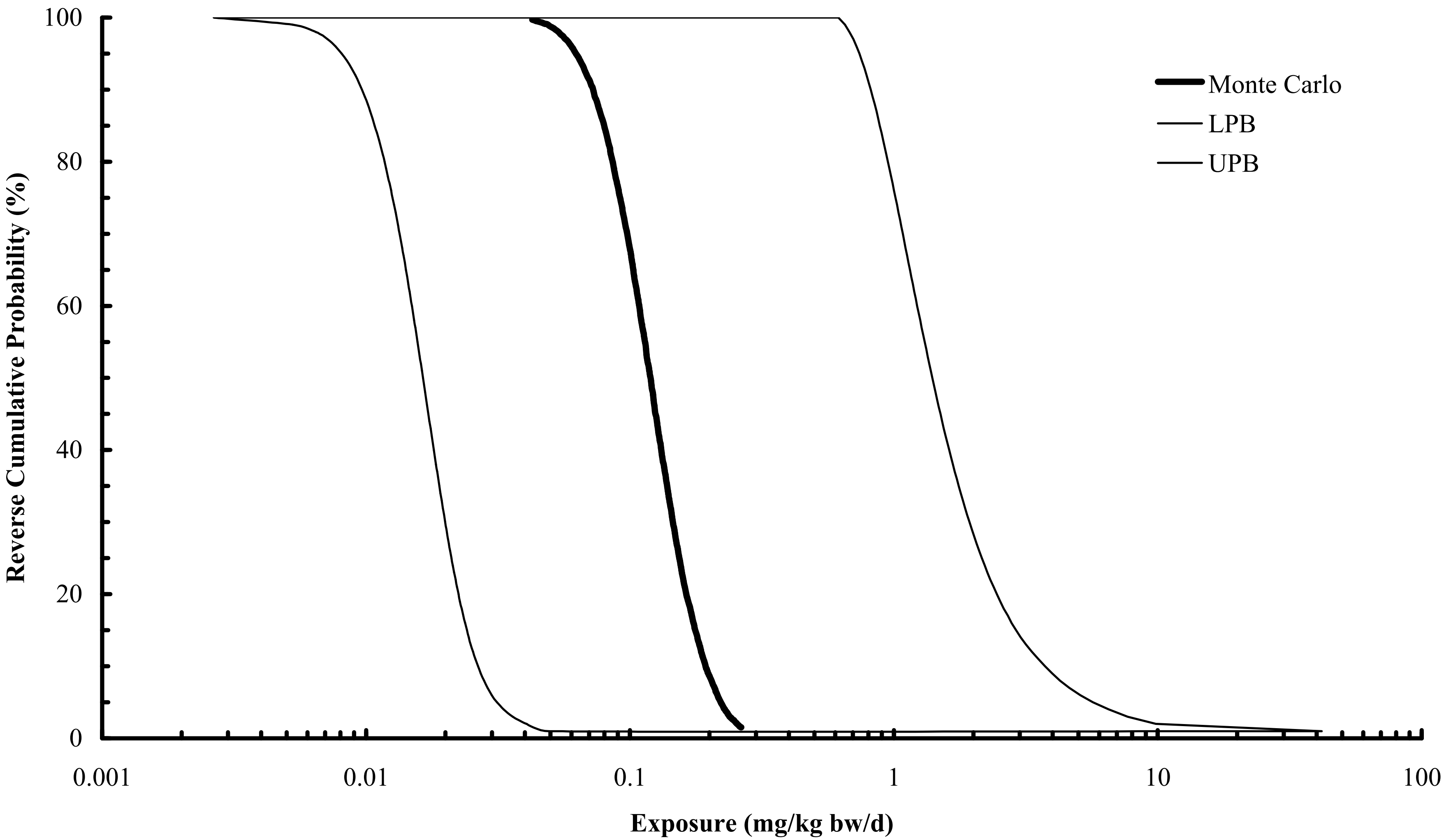

Figure H1-27. Reverse cumulative probability distribution of total daily intake rates of selenium by small sediment-probing birds in the reference areas, Calcasieu Estuary.

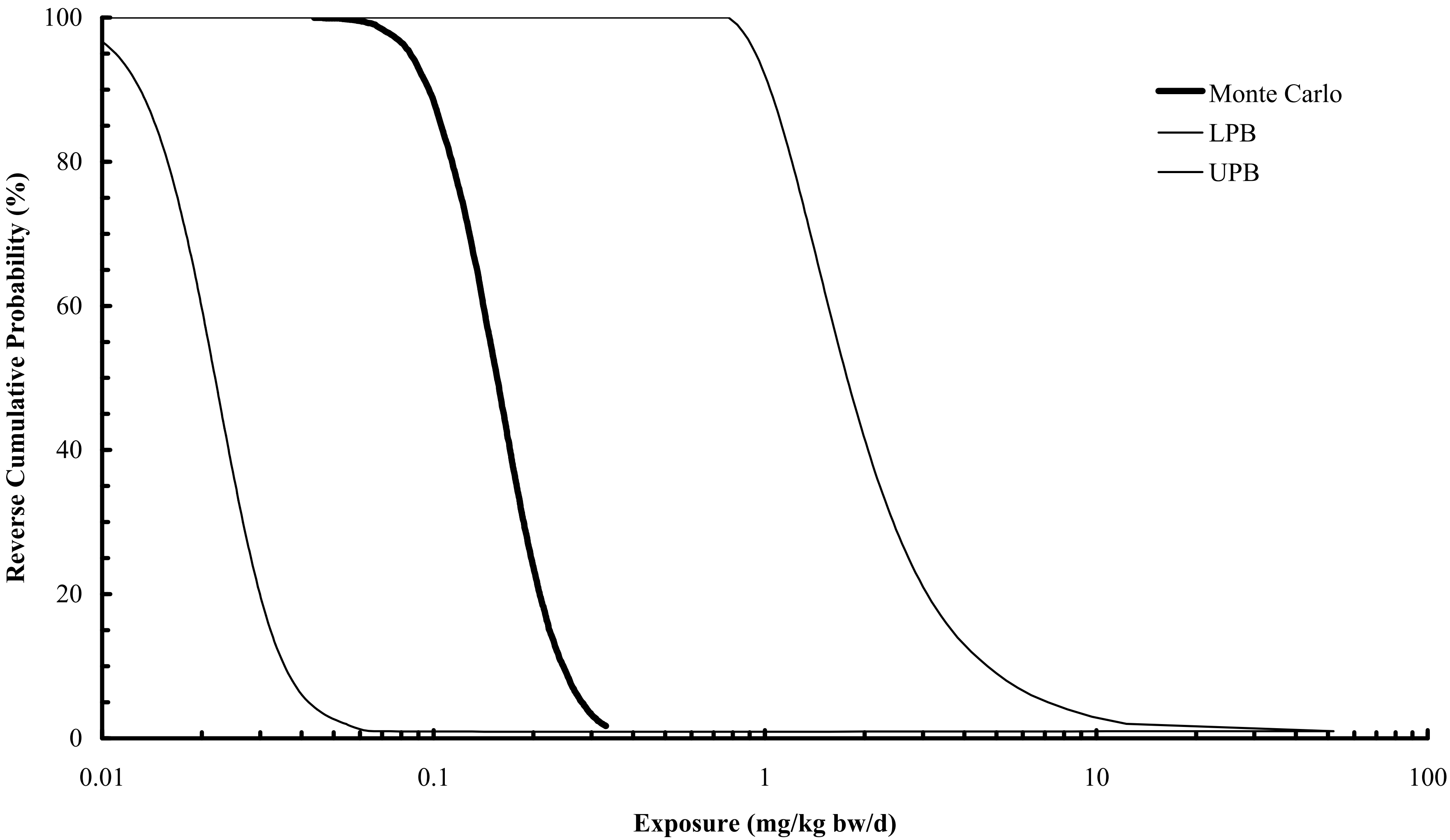

Figure H1-28. Reverse cumulative probability distribution of total daily intake rates of TCDD-TEQs by average-sized sediment-probing birds in Bayou d'Inde, Calcasieu Estuary.

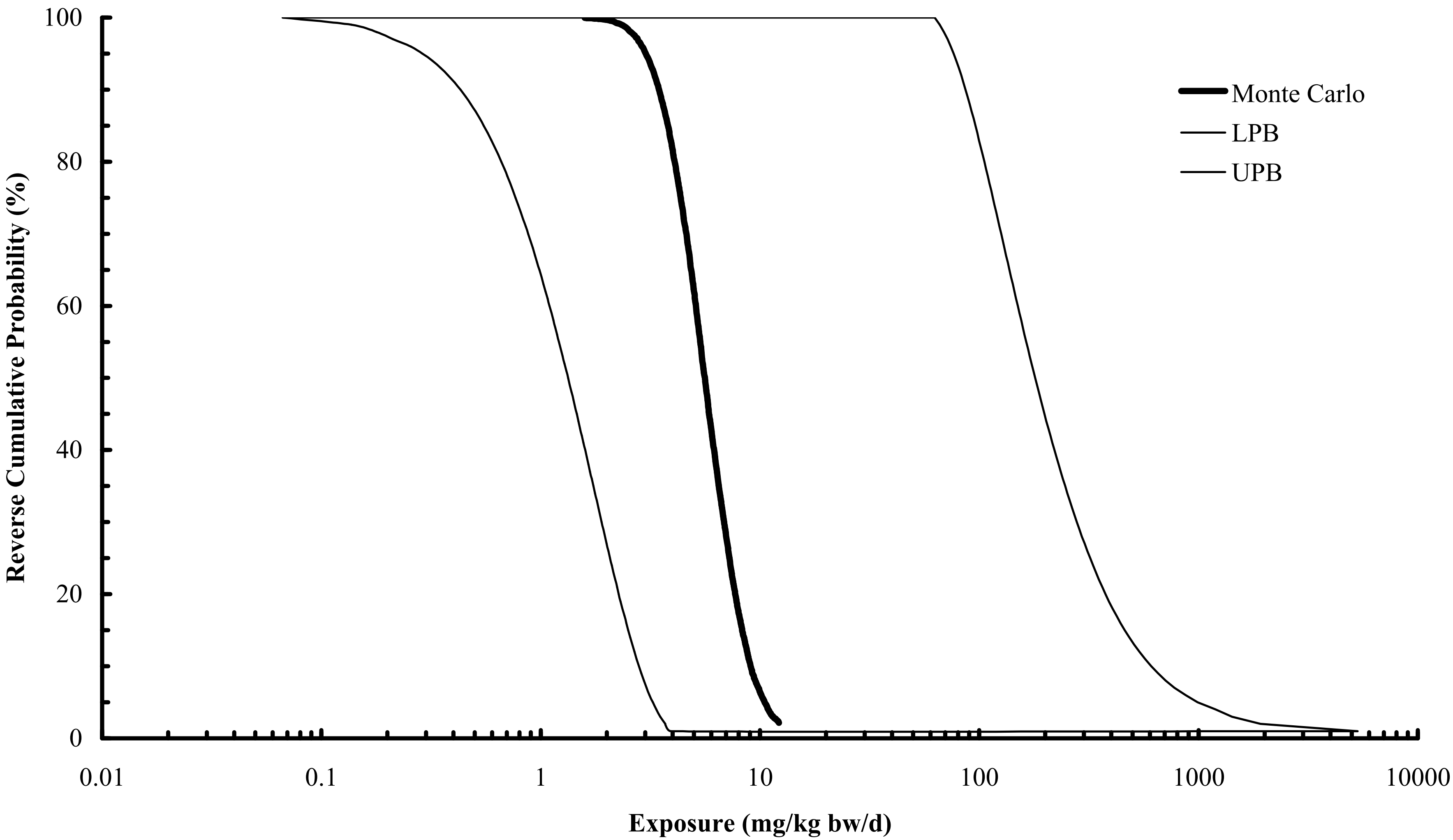

Figure H1-29. Reverse cumulative probability distribution of total daily intake rates of TCDD-TEQs by small sediment-probing birds in Bayou d'Inde, Calcasieu Estuary.

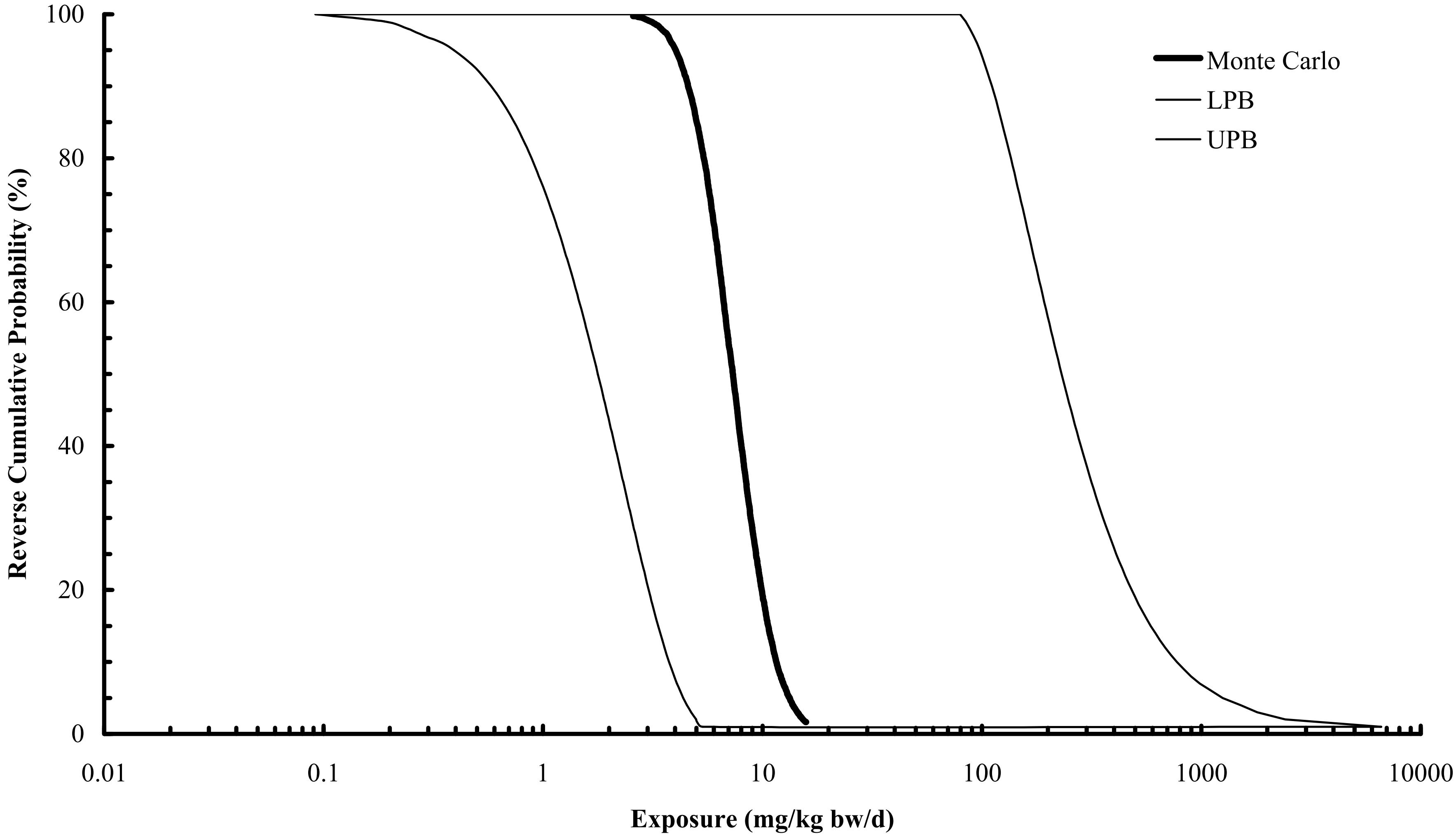

Figure H1-30. Reverse cumulative probability distribution of total daily intake rates of TCDD-TEQs by average-sized sediment-probing birds in the Upper Calcasieu River, Calcasieu Estuary.

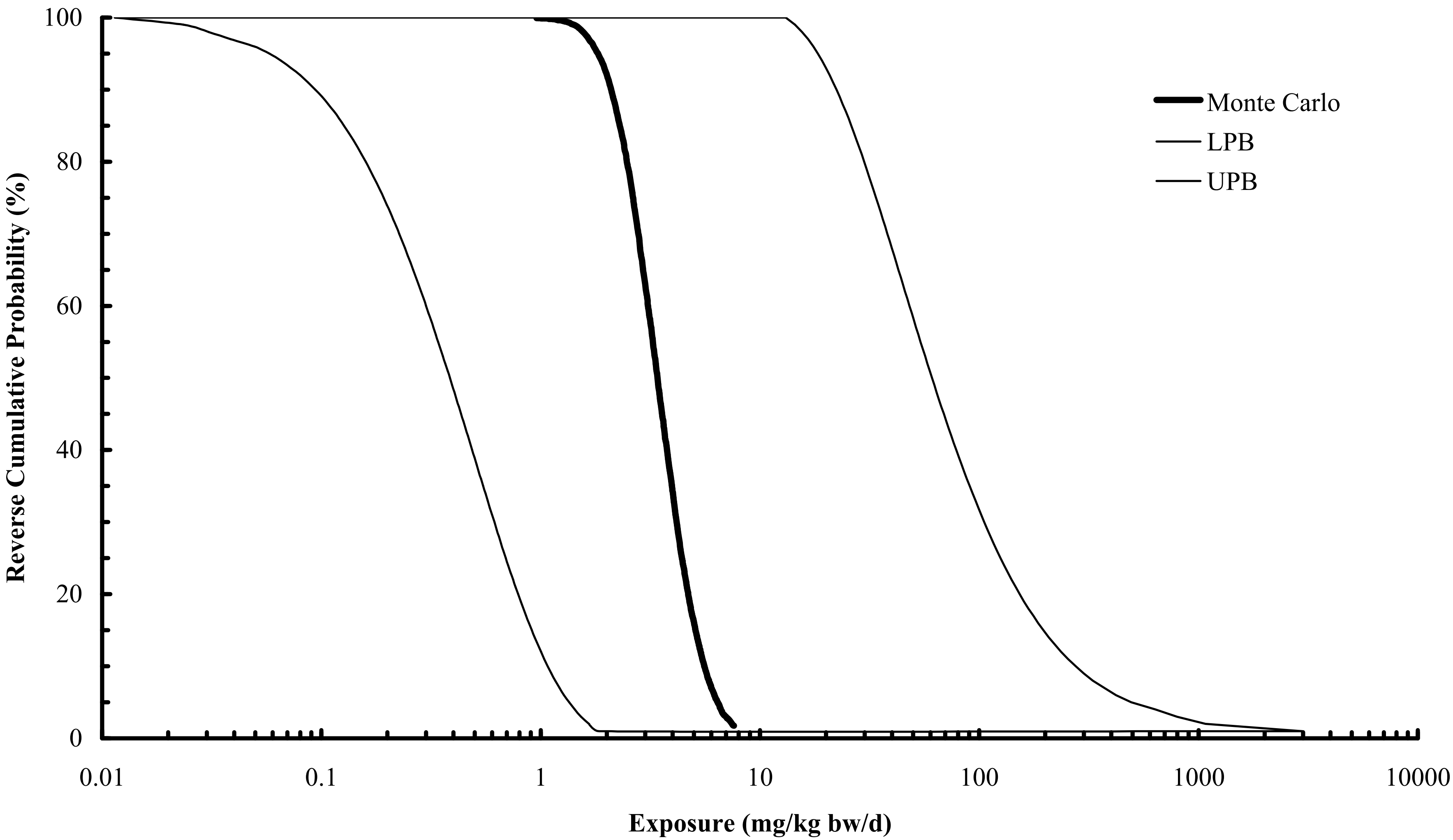

Figure H1-31. Reverse cumulative probability distribution of total daily intake rates of TCDD-TEQs by small sediment-probing birds in the Upper Calcasieu River, Calcasieu Estuary.

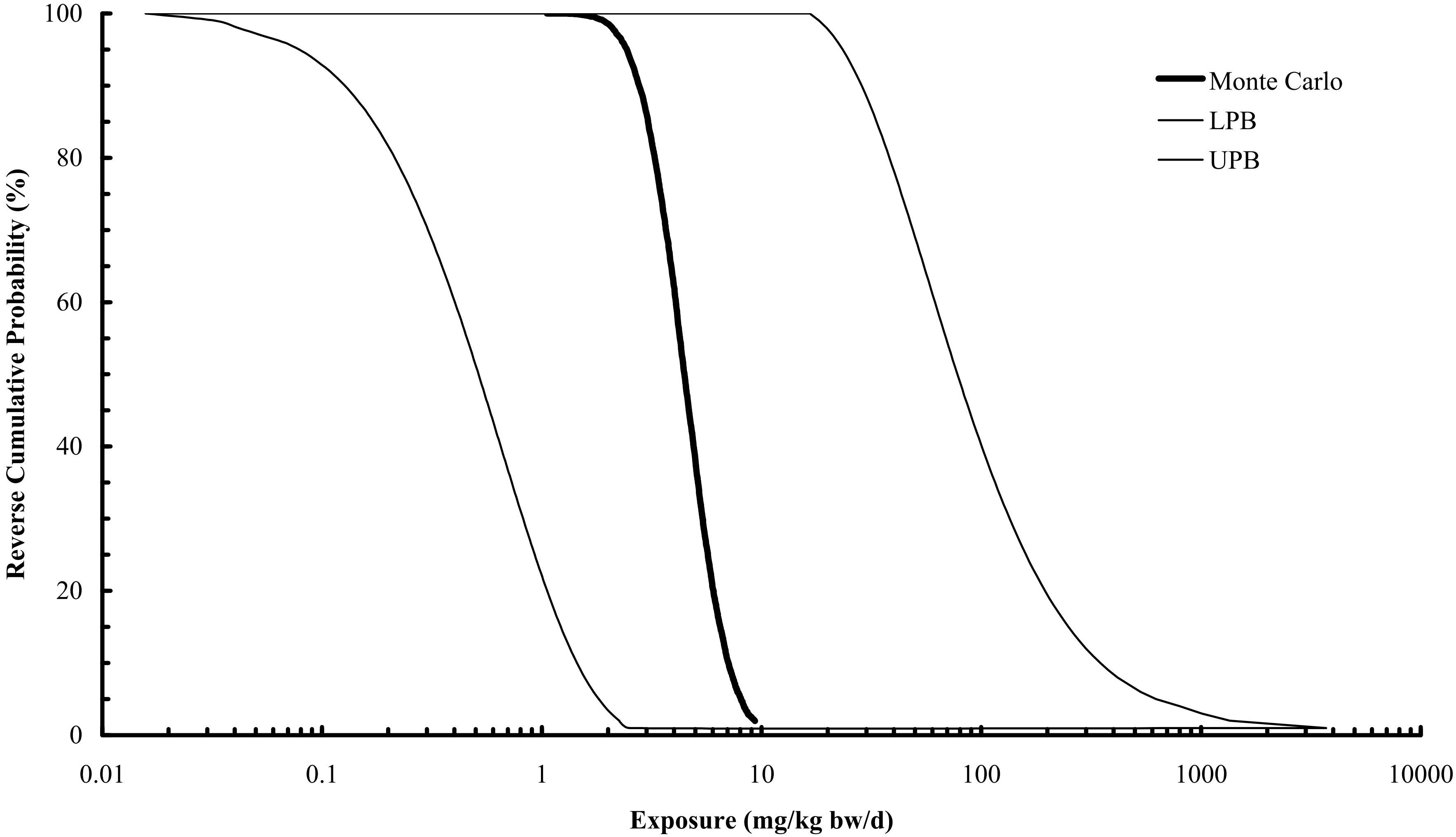

Figure H1-32. Reverse cumulative probability distribution of total daily intake rates of TCDD-TEQs by average-sized sediment-probing birds in the reference areas, Calcasieu Estuary.

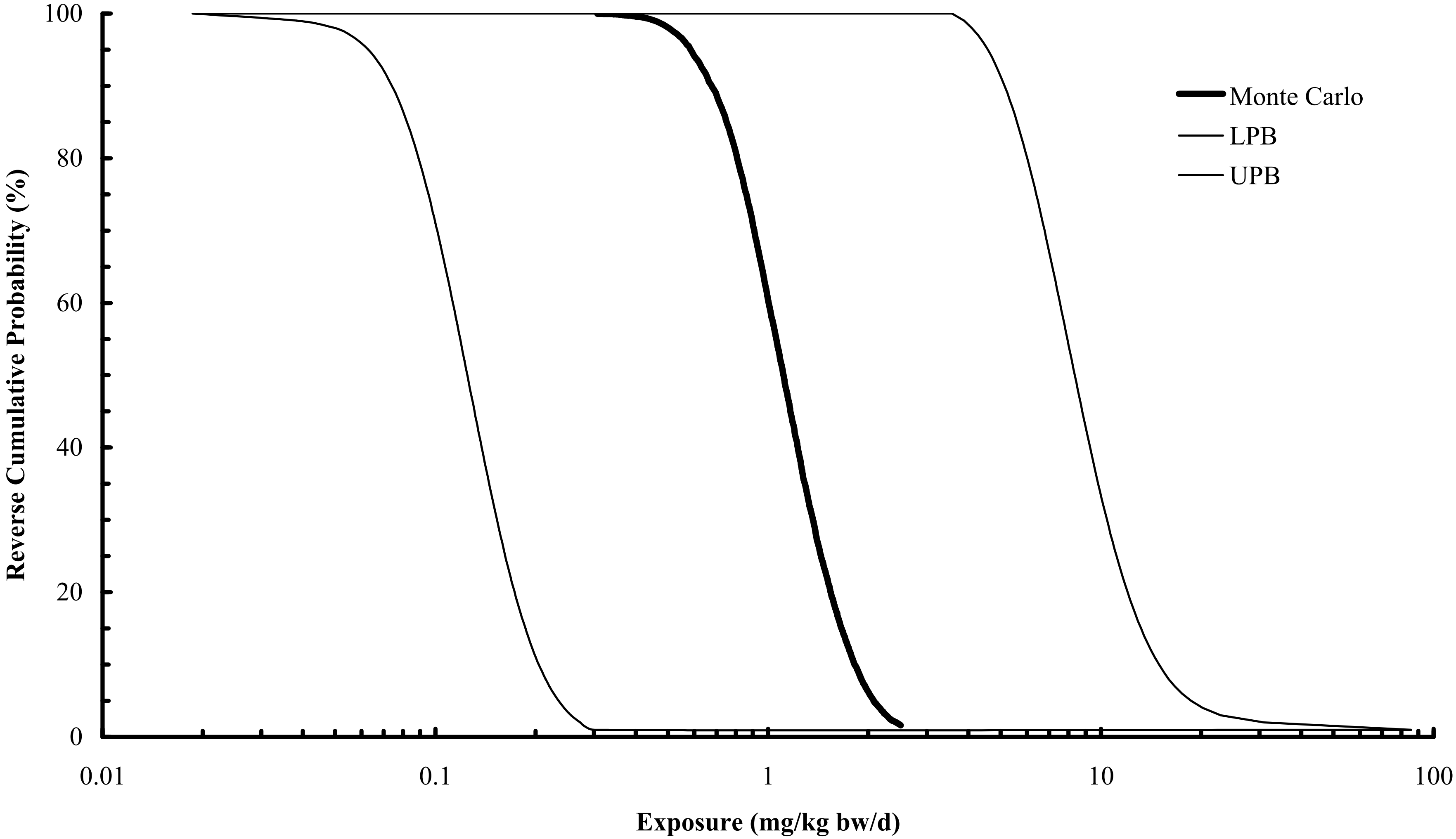

Figure H1-33. Reverse cumulative probability distribution of total daily intake rates of TCDD-TEQs by small sediment-probing birds in the reference areas, Calcasieu Estuary.